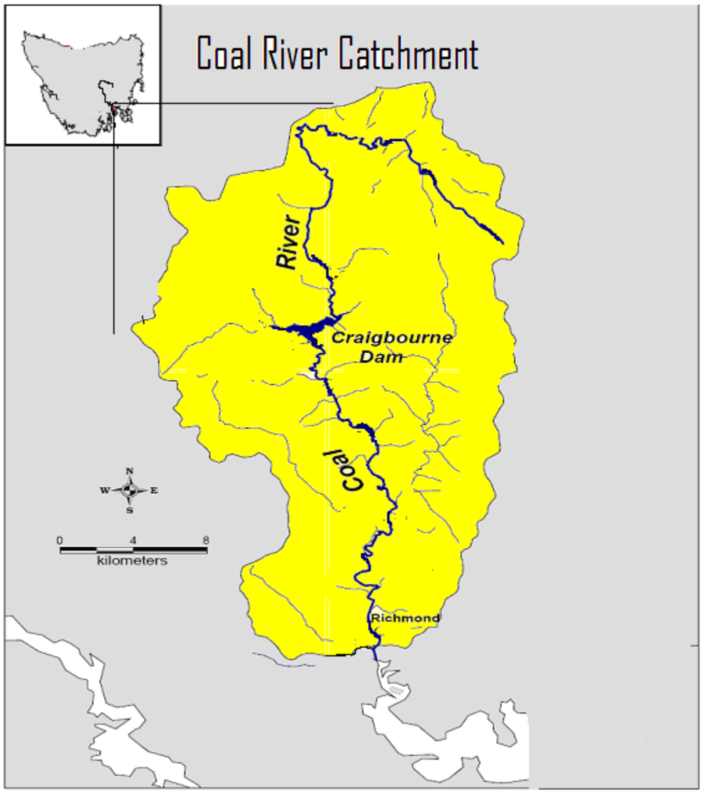
Quantifying the effects of land management interventions on water quality in the Coal River Catchment



By

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Submitted in fulfilment of the requirement for the degree of Masters in Agricultural Science

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Declaration

I declare that this thesis has not already been submitted for any other degree diploma in any other tertiary institution. I also certify that any help I have received, and all sources used, have been acknowledged in the thesis. To the best of the author's knowledge and belief, the thesis contains no material previously published or written by others person, except where due reference is made.

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Suresh Babu Panta

June 2011

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Abbreviations

ANOVA	Analysis Of Variance
ANZECC	Australian and New Zealand Environment and Conservation Council
DNR	Department of Natural Resources
DPIPWE	Department of Primary Industries, Parks, Water and Environment
DPIW	Department of Primary Industries and Water
DPIWE	Department of Primary Industries, Water and Environment
GIS	Geographic Information Centre
NSWEPA	New South Wales Environmental Protection Agency
NTU	Nephelometric Turbidity Unit
UNESCO	United Nations educational, Scientific and Cultural Organization
USGS	United States Geological Survey

Principle Units

km	kilometres
km^2	square kilometres
mm	millimetre
m	meters
cumecs	cubic meters per second
^0C	degree Celsius
ha	hectare
%	percent
mL	millilitre
ML	Megalitre
µS/cm	micro Siemens per centimetre

Abstract

This study examined spatial variation in water quality and its relationship to riparian land management in the Coal River Valley, SE Tasmania. Historical water quality data from stations at Baden, downstream of the Craigbourne Dam, Richmond and White Kangaroo Rivulet collected between 1999 and 2008 were obtained from DPIPWE. The water quality variables selected for study were water temperature, electrical conductivity (EC), dissolved oxygen (DO), water pH, turbidity, nitrate, total nitrogen (TN), dissolved reactive phosphorus (DRP), total phosphorus (TP) and stream flow. Riparian land use within one kilometre of the river was assessed and digitised using 2005/7 colour aerial photographs and water quality data for that period were examined and possible linkages investigated. The historical data demonstrates complex spatial patterns of in-stream water quality parameters in the Coal River Valley. The Craigbourne reservoir and apparent differences in geology between subcatchments explained some of the differences observed in water quality parameters better than land use. There was no significant variation in water temperature but higher values were recorded at Richmond. Similarly, higher EC was observed at the base of the catchment (Richmond) than at the top of the catchment (Baden). The influence of the Craigbourne Dam was reflected in higher pH, DO, DRP, TP and water flow at the station downstream of this large water reservoir in the middle of the catchment. In the White Kangaroo tributary higher nitrate and TN were observed but at Richmond, below its confluence with the Coal River, lower TP and TN were recorded. There was a significant negative correlation between DO and water temperature observed in the Coal River. However, positive correlations were found between stream flow and rainfall with turbidity at all stations except downstream of the reservoir. Stream nitrogen and phosphorus showed a significant relationship with rainfall at Richmond. Positive correlations of turbidity with nitrate, TN, DRP and TP show nutrients bound to sediment are a likely source of many nutrients in the river. Consequently, riparian vegetation could play a vital role in reducing sediment load and nutrient concentration in the river system. Subcatchment riparian land use and water quality data from 2005/7 suggests that lower turbidity at Richmond (2.82 NTU) compared to White Kangaroo Rivulet (4.25 NTU) and Baden (4.70 NTU) may be due to

the impact of higher percentages of willow trees in the Richmond subcatchment (1.92 %) followed by White Kangaroo subcatchment (0.26%) and Baden (0%). Riparian land management works such as planting native vegetation and fencing (approx 4 km) on the river banks could have reduced the sediment load and nutrient in the river by preventing erosion caused by stock access to river water. This is supported by 2005/7 nutrient data at Richmond where TP (0.015 mg/L) and TN (0.625 mg/L) were observed as compared to Baden (TP = 0.020 mg/L, TN = 0.69 mg/L) and Downstream Craigbourne Dam (TP = 0.022 mg/L, TN = 0.71 mg/L). However this lower part of the river still had the highest amount of willows in the riparian strip, despite willow removal programmes, and so the results are thus confounded. A higher percentage of native pastures in the riparian strip were found to be associated with lower turbidity in the river. Finally, forest cover was found to be positively correlated with nitrate nitrogen which is likely to be caused by nitrogen fixing acacia dominated forests.

1. Introduction

The Australian landscape has been changing continually due to agricultural and urban development since European settlement 200 years ago. This change has resulted in many problems in environmental sectors such as deterioration in the condition of many rivers. The degraded water quality in major river systems suggests that the current usage of resources is not sustainable and will continue to deteriorate if there is not judicial management of natural resources. Information on the condition of water quality and potential pollution sources is very important for the development and implementation of sustainable water use strategies to maintain a sound and healthy environment (Crosa *et al.* 2006; Zhou *et al.* 2007). Rivers and streams provide water for drinking, farming and other agricultural and non agricultural uses in both urban and rural communities so healthy water systems and their management crucial. Suitability of river water for various purposes, for instance, irrigation can be determined by evaluating the physico-chemical parameters and their geographical location. The major physico-chemical parameters that decide the suitability of water for irrigation are pH, EC, TDS, nitrate, sodium, potassium (Sundaray *et al.* 2009). Water quality and quantity is influenced by climate, geography and human interventions and is commonly highly variable spatially and temporally (Raj & Azeez 2009). So understanding the impact of human-environment interactions on water quality is essential if policies are to be developed to prevent further degradation (Singh & Singh 2007).

The worldwide deterioration of water quality in river and streams has been attributed to the direct and indirect contributions of natural processes and anthropogenic activities, including hydrology, climate, precipitation, agricultural land use practices and urban sewage management (Ravichandran 2003; Gantidis *et al.* 2007). Rivers are under continuous pressure due to various anthropogenic processes to meet the demand of increase global population (Singh & Singh 2007). The main contributors to this condition are land clearance for intensive farming, over-grazing of pastures on sloping lands, excessive use of agricultural chemicals such as fertiliser and pesticides, removal of riparian vegetation and direct drainage of urban and industry sewage into rivers.

Measures that minimise water pollution include improved land management practices such as conservation tillage on sloping lands to reduce soil erosion and leaching of nutrients and chemicals to the rivers. Establishment of sewage treatment plants can control urban and industry waste and reduce point source pollution in river systems (Sarkar *et al*. 2007). Maintaining vegetated buffers strips on the river banks is also considered important because they can reduce the input of soil particulates into the river. Temporal and spatial investigation of water quality in a watershed is important due to the seasonal and regional variation in water quality (Ouyang *et al.* 2006; Sundaray *et al*. 2006). Pillsbury & Byrne (2007) and Kannel *et al.* (2008) have reported on spatio-temporal variation in river water quality due to the impact of anthropogenic activities and the influence of natural processes. Similarly, Bu *et al*. (2010) reported that river water quality progressively degraded downstream from the point of origin.

Several water quality monitoring programs have been started at local and national levels in Australia due to increased public concern with the degrading water quality in rivers (Brainwood *et al.* 2004).

The aims of this study were to examine the effects of land use within the riparian zone on water quality in the Coal River catchment. The Coal River flows through a diverse and highly productive agricultural landscape in southern Tasmania. It drains a catchment area of about 540 km^2 and includes the urban settlement of Richmond on the banks of the river. The river serves as a major source of agricultural water supply and receives the agricultural runoff and urban drainage.

This catchment was chosen because of the diversity of land uses, the extensive work by landholders and the Coal Valley Landcare group to improve management of the riparian zone (through removal of exotic willows (*Salix* sp.), fencing to control stock access and establishment of endemic vegetation) and the availability of water quality data from four gauging stations. Willows influence stream condition through shading, reduced water flow and alteration of water channel morphology from hanging branches and roots

(Lisson *et al.* 1997). Their removal is commonly practiced as a means of improving water flow and quality.

While there is general consensus that the impact of the human intervention on the landscape since European settlement in Australia is a growing problem and threatens the sustainability of agriculture, water supplies and nature conservation, research at catchment scale in Tasmania has been limited. This study will make use of historic aerial photography and ground truthing to quantify the extent and type of land use and land management, and compare this with water quality data collected by the Department of Primary Industries, Parks, Water and Environment over the last nine years at gauging stations on the Coal River between Baden and the town of Richmond in South Eastern Tasmania, Australia.

The project aims to examine the relationship between land use and land management practices in the riparian zone and water quality in the Coal River by asking the following questions and testing two key hypotheses;

1) Is it possible to detect spatial variation in water quality in the Coal River as measured at DPIPWE gauging stations?
2) Is it possible to detect the impact of land use and land management within the riparian zone on water quality in the Coal River as measured at DPIPWE gauging stations?

Hypothesis

H_1 = It will be possible to detect a correlation between land use and land management practice within the riparian zone of the Coal River and water quality data collected among different gauging stations during 2005/7.

H_0 = It will not be possible to detect a correlation between land use and land management practice within the riparian zone of the Coal River and water quality data collected among different gauging stations during 2005/7.

2. Literature review

In this study the majority of the literature was derived from North American and European studies although there are significant climatic and environmental differences. But these studies do provide ideas how the river water quality can deteriorate and why. In addition there were few published studies available for Australian conditions related to the land use affects on water quality at a catchment scale or on a riparian zone basis. For example studies conducted in Rous River catchment in northern NSW Australia showed that elevated levels of nutrients were associated with leaching of excess fertiliser that have been applied in cane land (Eyre & Pepperell 1999). Ierodiacanou *et al.* (2005) studied a regional scale assessment of land use change on nutrient exports by using an export coefficient model, remote sensing and GIS technique in south west Victoria. During period of 1980 to 2002 the modelled phosphorus and nitrogen loads were increased by 0.14 kg/ha and 1.37 kg/ha respectively when land use changed from dryland pasture to more intensive agricultural activities such as cropping and irrigated pasture. Similarly, empirical studies have been done on the significant contribution of agricultural land use (Nash *et al.* 2004; Webster *et al*. 2001) and managed pasture land (Nash & Halliwell 2000; Fleming & Cox 2001) to excessive phosphorus concentration in the waterways of South Australia. However, some review papers related to land use and nutrient export in river systems have been written from an Australian prospective but these often use northern hemisphere data due to lack of relevant long-term Australian data sets (Young *et al*. 1996).

2. 1 Water quality

Water quality refers to the chemical and physical characteristics of river water which enables it to maintain healthy aquatic life and meet human needs. River water quality can be influenced by several factors including the lithology of a watershed, climatic conditions, and atmospheric and anthropogenic inputs (Bellos & Sawidis 2005). Water quality is mainly affected by soil erosion in agriculturally–dominated catchments and concentrations of nitrogen and phosphorus are important parameters describing the quality of water (Mattikalli & Richards 1996). Higher levels of nitrogen and phosphorus in rivers or dams can cause eutrophication which degrades surface water quality especially by reducing levels of dissolved oxygen at night (Cooper 1993). Other potential sources of the nutrients in the rivers are wastewater treatment, industrial discharges, agricultural fertiliser, livestock manure and atmospheric deposition (Carpenter *et al*. 1998). Higher amounts of nitrate- nitrogen and ammonia- nitrogen can be toxic to the aquatic life as well as animals (Kumar 1998). The presence of salinity in the Coal River is considered in part a natural phenomenon, but some areas show high and increasing levels (>1500 μS/cm) of surface water electrical conductivity above the level recommended for Tasmanian rivers (ANZECC 2000). Bedrock geology can also affect pH and conductivity (Silsbee & Larson 1982). During the summer season the Coal River is converted to a series of disconnected ponds and stagnant pools due to low flow which in turn causes elevated conductivity, turbidity, nutrient concentrations, and depleted oxygen levels (DPIPWE 2003b). Water quality not only affects agricultural industries but also has significant influence on tourism and fishing. Additionally, it has indirect effects on the economy of the state as low water quality increases the water treatment costs for both domestic and commercial use. Moreover it causes imbalance in aquatic ecosystems and adversely impacts biodiversity and environment. Some of the Coal River water quality parameters are described as below:

2. 1. 1 Water temperature

Optimum stream temperature is vital for the aquatic life and it has biological, chemical and ecological impacts on the river system (Barton *et al*. 1985; Schlosser 1991; Stott & Marks 2000). Temperature is an important physical characteristic of water (Webb *et al.* 2008) and has been considered a major regulator of the aquatic living system that determined the geographical distribution, growth rate and survival of fish species and other aquatic organisms (Holmes & Regier 1990). Water temperature affects the rate of in-stream chemical reactions (Feller 1981) particularly high temperature can increase the speed of biological processes in aquatic plants and animals by depleting oxygen levels in water (Mayo & Noike 1996). Stream temperature influences the concentration of dissolved oxygen in the water (Davis 1975) which affects respiration and metabolism of aquatic life (Eckert 1988). Many factors affect the stream temperature for instance upstream land use activities (Stott & Marks 2000), riparian tree removal along water courses (Brown & Krygier 1970; Martin *et al.* 1985; Beschta & Taylor 1988) and forest harvesting activities outside the buffer zone (Bourque & Pomeroy 2001). In addition, season and altitude have profound effect on the stream water temperatures. During the summer time water temperature in the river may reach higher values where there is a lack of riparian vegetation cover and it tends to be cooler in areas that are well shaded. The upper catchments areas higher in altitude are generally cooler than the lower catchment. It can be predicted that not only climate change but also multiple anthropogenic activities will have significant impact on streams in the future. For example, increased watershed imperviousness and reduction of the riparian vegetation during urbanization will alter water temperature in the river (Nelson & Palmer 2007).

2. 1. 2 Dissolved oxygen

Dissolved oxygen (DO) is needed for the aquatic plants and animals (Zweig *et al.* 1999) and may vary in the water system due to temperature, salinity, biological activity and rate of transfer from the atmosphere. Its concentration mainly depends on the salinity and temperature of the water. In general, cold water with low salinity level has a higher concentration of dissolved oxygen. Due to in-stream photosynthetic and respiratory activity of aquatic plants, dissolved oxygen concentration is lower at night than during daylight hours (ANZECC 2000). During decomposition of biodegradable organic substances the concentration of dissolved oxygen is reduced because oxygen is used by bacteria for this process (Svobadova *et al*. 1993). The water in dams or reservoirs has higher fluctuations in DO concentration than the sea or high speed running waters and the lowest DO occurs in the morning and higher concentrations in the late afternoon (Boyd 1990). Chessman & Robinson (1987) reported that prolonged drought reduced the dissolved oxygen to 2 g/m^3 in the lower LaTrobe River, Victoria which is below the acceptable range of >5 g/m^3 needed to support a diverse aquatic population in the river (ANZECC 2000).

2. 1. 3 Turbidity

Turbidity is an expression of the clarity of water. Water becomes turbid due to presence of suspended materials. There are several factors responsible for changes in turbidity including suspended material such as clay, silt, finely divided organic matter, soluble compounds, planktonic species and microscopic organisms, variable rainfall patterns and stream flow (DPIPWE 2003b; NLWRA 2001).

Turbidity is identified as the most widespread water quality issue in Australian rivers and streams (NLWRA 2001). The turbidity in the river and dam water is significantly influenced by anthropogenic activities, associated with changes in land cover and land use patterns in the catchment. Land use practices in the catchment are thought to be the major cause of the various water quality problems in the river (DPIPWE 2003b). Turbidity affects light penetration which in turn affects ecological processes that depends on sunlight. Moreover, light penetration also affects the temperature regime of surface waters.

In the Coal River catchment erosion from river bank and surrounding paddocks, direct access of stocks into the river and change in riparian vegetation are the main sources for increased turbidity in river. Delivery of sediments and nutrients especially nitrogen and phosphorus to the stream and their storage for extended periods in Craigbourne Dam has caused large blooms of blue-green algae which are becoming a problem for the health of the Coal River system (DPIWE 2003b). The blockage of irrigation equipment by blue green algae can result in uneven flow and increase maintenance costs (ANZECC 2000). Blue green algae (*Cynobacteria* species) can produce toxins and water with high levels of these algal used to irrigate lettuce and cabbages may create a potential health risk for human consumers (Jones *et al.* 1993).

2. 1. 4 Salinity (electrical conductivity)

Salinity of streams and rivers is affected by the parent materials from which soil is formed and is another major water quality issue in Australian rivers. In general Australian surface waters are naturally highly saline. Salinity in surface water refers to the dissolved concentration of salts such as sodium chloride (NaCl), magnesium chloride ($MgCl_2$). Stream flow may dilute salt concentration for example during periods of high flow, and ground water flow can influence conductivity where ground water is naturally saline. Low summer flows are often characterised by high levels of conductivity due to ground water domination. During higher winter flows the lower saline surface waters dominate, resulting in a decrease in in-stream conductivity. High levels of soluble salt can reduce the plant productivity and kill plants. To assess salinity, electrical conductivity is used to estimate the concentration of total dissolved salts in water (ANZECC 2000). Salt in river water originates from dissolution weathering of rocks and soil minerals and application of this saline water in crop production causes salinity problems in the soil (Sundaray *et al.* 2009). The effect of salinity and its concentration and sensitivity to the plants are presented in Table 1 below.

Table 1 Irrigation water suitability rating based on the electrical conductivity.

EC(μS/cm)	Water salinity rating	Plant suitability [a]
<650	Very low	Sensitive crops (turnip)
650- 1300	Low	Moderately sensitivity crops (spinach, cabbage, celery)
1300- 2900	Medium	Moderately tolerant crops (olive, garden beet)
2900-5200	High	Tolerant crops(canola, wheat, oat, perennial rye grass)
5200-8100	Very high	Very tolerant crops (barley)
>8100	Extreme	Too saline (puccinellia)

Adapted from DNR (1997), [a] DPI NSW (2006)

2. 1. 5 Water pH

The pH refers to the concentration of hydrogen ions (H^+) in water. In streams, the pH value is influenced by several factors such as natural underlying geology, soil chemistry, flow characteristics, vegetation and land use practices (Bobbi 1999a, b). The pH or acidity of Tasmanian streams is generally in the range 5.5 - 7.5 but in some humic rich lakes and rivers the pH range is 4.0 to 6.5 (ANZECC 2000). The pH range is variable both seasonally and diurnally depending upon environmental conditions and biological and atmospheric processes (UNESCO 1992). A pH of more than 8.3 can point to the existence of bicarbonate, carbonate and sodium in the water bodies (ANZECC 2000). The use of alkaline water in irrigation can reduce the availability of trace elements which can in turn affect plant growth (Slattery *et al.* 1999) while irrigation water with pH < 5 can cause corrosion in the irrigation distribution system (Gill 1986). Moreover, low pH values decrease the availability and amount of dissolved inorganic phosphorus in the water (Zweig *et al.* 1999). The normal pH range of irrigation water should be from 6.5 to 8.4 and outside this range can cause nutritional imbalances in plants (Sundaray *et al.* 2009).

2. 1. 6 Nutrients

In Australia, nutrient concentrations in rivers and streams are a major water quality issue. It seems that agricultural and urban disturbance within a catchment generally leads to increases in nutrients exported to river systems. In water, nitrogen can be found in different forms such as ammonia (NH_3), ammonium (NH_4^+), nitrite (NO_2^-) and nitrate (NO_3^-) while phosphorus is also found in different forms such as dissolved inorganic phosphate or colloidal phosphate (ANZECC 2000). Of these nutrients, increased levels of nitrate not only play a vital role in eutrophication of rivers or pond water (NSWEPA 1995) but also contribute to episodic acidification of surface water (Wellington & Driscoll 2004). Nitrogen is an essential nutrient for plant growth however high levels of nitrogen in irrigation water can be occur through natural rock types and applied chemical fertilizer. Nitrogen concentrations of more than 5 mg/L in irrigation water can affect some sensitive plants (Sundaray *et al.* 2009). It is well reported that environmentally significant concentrations of phosphorus (>0.05 mg/L) can cause algal bloom (ANZECC 1992; Foy & Withers 1995). Similarly the two most important pathways for nutrient loss from the landscape to rivers appear to be leaching, erosion and surface run-off. Increased nutrient loads in rivers can boost the production of algae which affects stream turbidity and flow regimes. Nutrients, particularly phosphorus, are transported in different forms such as dissolved and particulate or sediment-attached form (Kirkby *et al* 1997; Nash & Murdoch 1997; Stevens *et al.* 1999). Nutrient concentrations in the river are affected by several factors. For example, the concentration and forms of nitrogen differ according to geography (Rohm *et al*. 2002), type of vegetations cover (Lovett *et al*. 2000; Lewis & Likens 2000; Binkley *et al.* 2004) and geology (Holloway & Dahlgren 1999). Holloway *et al.* (1998) reported that metasedimentary and metavolcanic rocks were the source of higher nitrate (NO_3^-) concentrations in lower reaches in the Mokelumme River watershed in Central Sierra Nevada of California. Similarly, Wooten *et al.* (1999) found that that rivers draining though limestone bedrock had higher nitrate (NO_3^-) concentration than those draining sandstone bedrock. Binkley *et al.* (2004) described lower phosphate concentrations in streams draining through igneous bedrock and higher concentrations in volcanic bedrock or glacial till parent materials respectively. Forest defoliation by insects may increase nitrate (NO_3^-) concentration in the adjacent river water (Swank *et al.*

1981; Eshleman *et al*. 2000). Meynendoncks *et al.* (2006) reported that river nitrate concentrations were positively correlated with effluent coming from the wastewater treatment plants and agricultural land while phosphorus concentrations were influenced by industrial waste.

The effect of watershed land use on rivers nutrient concentrations has been studied over the past two decades by several researchers (Osborne & Wiley 1988; Wahl *et al.* 1997; Tufford *et al*. 1998). Brett *et al.* (2005) reported that phosphorus concentrations in the stream were moderately or strongly (R^2 = 0.58) correlated with land cover type. Agricultural and urban land use practices can increase nutrient concentration in the adjacent river water (Soranno *et al.* 1996; Carpenter *et al.* 1998). Tree type and the proportion of forest in a catchment can have a significant influence on nutrient concentrations in the rivers. Van-Miegroet *et al.* (1992) described that ammonium produce by red alder trees in summer was nitrified and stored in the forest soils and then washed out during winter season to the waterways when the biological uptake of inorganic nitrogen is at minimum levels. Algal blooms in water bodies have been reported as a result of diffuse agricultural sources of phosphorus (Correll 1998; Dils *et al.* 1999; Daniel *et al.* 1999; Stevens *et al.* 1999) and increased levels of nitrogen (Rabalais *et al.* 2002) which in turn cause hypoxia or dead zones for fish.

2. 1. 7 Water flow

Human intervention in river systems can have a significant impact on natural stream flow (Davies 2002). Some vegetation such as trees hanging down from the river banks may have a significant impact on the stream flow rate and aquatic environments, although they may help to prevent erosion of the river bank. In the Coal River, heavy infestations of willow trees on the river bank are likely to have had significant effects on stream flow. In general during winter and spring, base flows are higher and more continuous while in summer, when stream flow is low, they influence macro invertebrate populations in the river water. High summer temperatures and low summer flows have reduced the number of macro invertebrate in Buttons Creek in Tasmania and this is thought to be due to the deposition of suspended particulate matter in low flow conditions (Cotching & Sims 2003). Flow also has a significant impact on nutrient loads in the river, such as the work reported by Gökbulak *et al.* (2008) where nutrient loads were positively correlated with stream flow in an oak beech forested catchment in Turkey.

2. 2 Land use impacts on water quality

Human intervention can have significant impact on water quality through different types of land use activities. The different influences of land use practices in a catchment on the river system are shown in Figure 1. The major reasons for spatio-temporal variations in water chemistry in an Indian tropical river were found to be due to changes in land use practices and the formation of dams in the rivers (Raj & Azeez 2009). Among several factors responsible for degrading river water quality, agricultural and urban land uses have been found to be major causes through polluting by sediments, nutrients, heavy metals and faecal bacteria (Doyle 2005). Sediments not only affect the habitat of aquatic organisms (Boulton *et al.* 1997; Jowett & Boustead 2001) and clog the gills and feeding mechanisms of fish (Wood & Armitage 1997; Eillis *et al.* 2002) but also reduce aquatic photosynthetic productivity (Ryan 1991). Nitrate-nitrogen present in stream water is widely used as an indicator of water quality degradation. Spalding & Exner (1993) reviewed the increasing evidence of nitrate concentrations in surface and subsurface water in different parts of the world and found the major sources of these nutrients to be chemical fertiliser and animal manure. When the nitrogen application rates on both intensive and hobby farms exceed the uptake rate of plants, streams and groundwater are likely to become polluted with nitrate and making it difficult to protect water quality (Meybeck & Helmer 1992; Harper *et al.* 1992). Nitrate is susceptible to leaching to waterways because of the small size and high mobility of the anion (Keeney 1986).

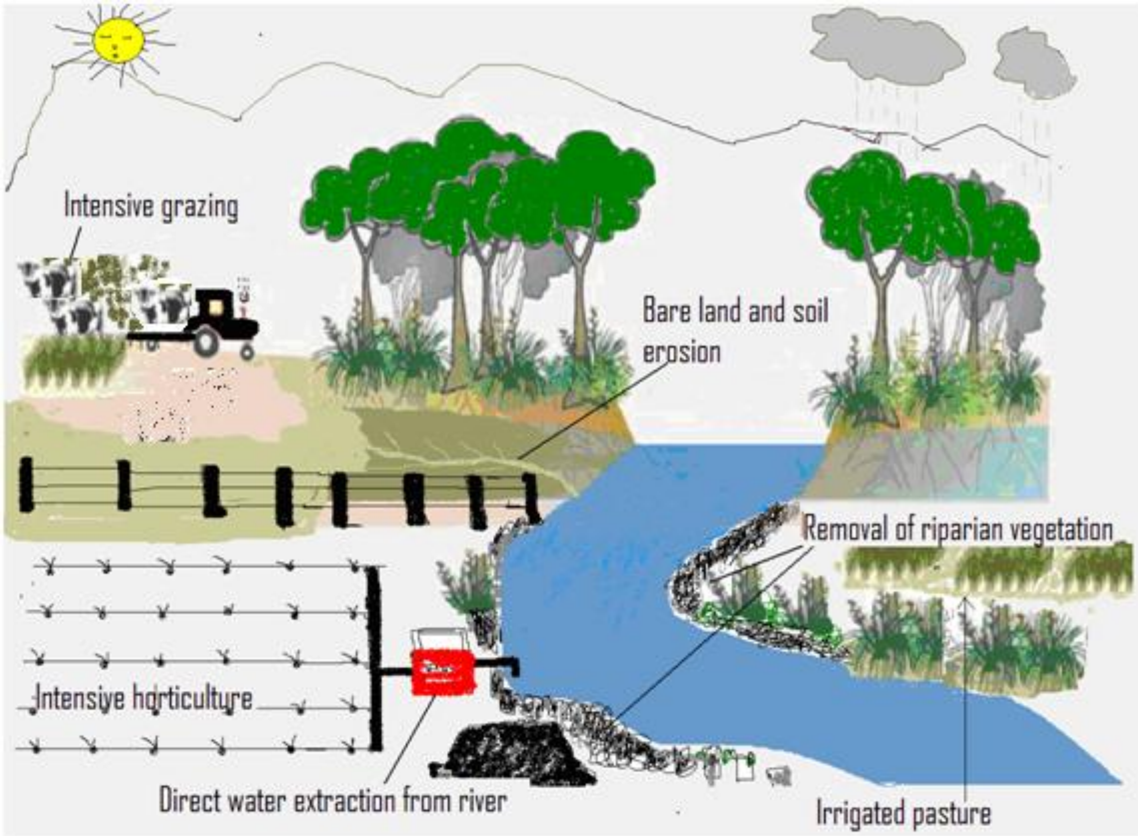


Figure 1 Conceptual diagram of the mechanisms leading to degraded water quality in river systems

Similarly, McColl (1978) reported a correlation between the amounts of fertiliser applied in fields with the concentration of nutrients in river water. The phosphorus concentration found in streams running through improved pasture was about 15 times higher than that of streams passing through a forested catchment (Cooper & Thompsen 1988). Some soil types also affect nutrient retention in agricultural lands. Intensive manure applications to small areas of land increased the potential for phosphorus movement to surface water system (Sims *et al.* 1998). During storm events, phosphorus can be transported from manure applied to medium and fine textured soils adjacent to river systems through runoff (Pote *et al.* 1999) while in coarse-textured soil, phosphorus can leach to groundwater and move laterally to other areas through subsurface flow (Novak *et al.* 2000). These processes of phosphorus movement have environmental consequences because the dissolved phosphorus is the most available form of phosphorus and taken up by algae and aquatic weeds (Sonzogni *et al.* 1982).

Tillage operations, forest logging or fire that causes loss of vegetation and exposed soil can lead to rill, sheet and gully erosion. Increasing pressure to cultivate on steeper marginal farmland to produce more food to feed an increasing population can have a significant impact on water quality as these land types commonly require more fertiliser than prime land (Charbonneau & Kondolf 1993) leading to increased nutrient movement to stream. Continual over grazing is another most important factor leading to soil erosion in rangelands (Myers *et al.* 1985). Use of chemical fertilizer, pesticides and soil disturbance in agriculture are major contributors to non point source (NPS) pollution of surface water quality. Tong & Chen (2002) found a significant relationship between land use and in-stream water quality with higher levels of nitrogen and phosphorus recorded in river water that drained from the agricultural land than urban land. Filoso *et al.* (2003) showed a significant positive correlation between nitrogen export and agricultural ($r = 0.75$) and urban land use ($r = 0.69$), but a negative correlation with pasture ($r = 0.60$) and forest ($r = 0.56$) land use types at $p < 0.05$. Similarly, other researchers reported that the use of nitrogenous fertilizer and the type of land use were related to the amount of nitrogen exported to the adjacent river (Little *et al*. 2003; Buck *et al*. 2004; Donner *et al.* 2004; Lattin *et al.* 2004; Woli *et al.* 2004). Densely populated catchments tend to export higher levels of nitrogen to rivers, most commonly due to sewage inputs (Jordan & Weller 1996).

Nutrients, especially phosphorous and nitrogen derived from agricultural soils to rivers, can cause eutrophication or abrupt algal growth and consequently increase water turbidity (Charbonneau & Kondolf 1993). Higher nitrogen in North American rivers was found to come from fertilizer applied to agricultural catchments (Boyer *et al*. 2002). Forests can also be a source of elevated nitrate concentrations in the river system. A study conducted in mixed conifer forests in Southern California showed that elevated nitrate concentrations in rivers were caused by nitrogen saturation where atmospheric deposition reached 20 to 25 kg N /ha/year or greater (Fenn *et al* 1996; Kiefer & Fenn 1997). Old-growth forests with high rates of deposition but reduced nitrogen demand and retention capacity tend to leach nitrogen to rivers (Fenn & Poth 1999). Higher concentrations of total nitrogen and total phosphorus were recorded in areas where a higher proportion of agricultural land use was in practiced (Lenat & Crowford 1994).

2. 3 Role and condition of riparian vegetation

Riparian land is defined as the narrow strip of land adjoining riverbanks, gullies and depressions, surrounding lakes and reservoirs, wetlands and river floodplains (Price & Lovett 2002). They are not only vital areas in the catchment, supporting high level of biodiversity, but also control flows of energy and nutrients between terrestrial and aquatic ecosystems (Naiman & Decamps 1997). Riparian lands are highly fertile areas frequently exposed to over use. In most cases these areas have been heavily cleared for intensive farming such as cropping and grazing. A conceptual diagram (Figure 2) illustrates current best management practice for riparian land. Vegetated strips on the river banks act as buffers to reduce runoff, intercept pollutants (Mickelson *et al.* 2003; Parkyn 2004) and are seen as important to regulate the movement of sediments and nutrients to rivers (Johnson *et al.* 1977; Omernik *et al.* 1981, Peterjohn & Correll 1984; Hill 1996, Tufford *et al.* 1998; Anbumozhi *et al.* 2005). Similarly, Lowrance *et al.* (1984) found that nitrogen concentrations were reduced by 65% through a riparian forest, due either to uptake by the plants or denitrification of sediment trapped on the riparian zone. It has been reported that well managed multi species vegetated riparian buffer strips such as fescue (*Festuca* species), switch grass (*Panicum virgatum*) and woody buffer can trap or reduce up to 70-80% of total nitrogen, 62 – 83% of nitrate- nitrogen (NO_3- N) and 73 - 78% total phosphorus loading to the adjacent river by intercepting the nutrients bound sediment on the riparian zone (Clausen *et al*. 2000; Lee *et al.* 2003). A study conducted in California, by Triska *et al*. (1993) found that the main process of nitrogen removal in riparian buffer strips was denitrification under anoxic conditions.

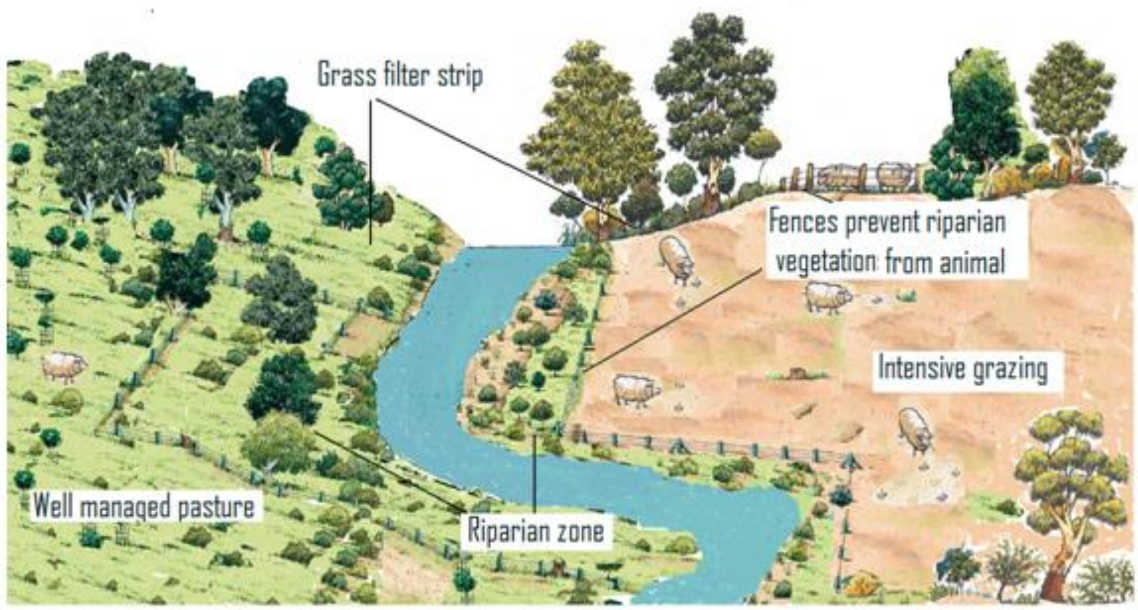


Figure 2 Conceptual diagram of riparian zone and role of vegetations.

In agricultural watersheds, uncultivated riparian strips can play an important role in reducing nutrients loading to the river (Cey *et al.* 1998) and also have a major influence on water quality and in-stream biological function through providing shade (Cumming 1993). Basnyat *et al*. (1999) reported that riparian land use was more significant in determining stream nutrient concentrations than land use over the whole river basin. During extreme climatic conditions such as heavy rainfall, well maintained riparian vegetation can act as a filter to reduce the amount of soil particles and nutrients moving from unprotected bare and cultivated fields into the river channel. These processes prevent deterioration of in-stream habitat by avoiding siltation (Price & Lovett 2002) and moderating diel temperature variation on the river (Broadmeadow *et al.* 2010). At the same time the vegetation helps to increase the aesthetic value of land (Karssies & Prosser 1999; Parkyn 2004). Rivers with less riparian vegetation cover is likely to have lower dissolved oxygen concentration (Wilcock *et al.* 1998) as oxygen solubility decreases as temperature increases. In summary, riparian vegetation is an important factor in the landscape which has a significant effect on streams and river systems (Cumming 1993).

Willows (*Salix spp*.) have now become a dominant component of riparian vegetation in southern Australia. They are an exotic tree species introduced to Australasia in the 19th century (Mitchell & Frankenberg 1993; Cremer *et al.* 1995). However, their rapid expansion in Tasmania river systems has only been noted in the last 50 years (Radcliffe 1990). This expansion was triggered when willows cuttings were planted to the river bank to reduce erosion after clearing the native plant for agriculture (Bobbi 1999). Ladson *et al.* (1997) reported that 28 willow species are present in Australia but three species are most abundant, namely *Salix fragilis* L. (Crack willow), *Salix babylonica* L. (Weeping willow) and *Salix alba* L. (White willow). Of these, S*alix fragilis* L. is wide spread in Tasmanian river systems (Cremer *et al.* 1995). These species have been categorised as notoriously fast invading shrubs are responsible for flooding and erosion by colonizing the river bed and bank (Meikle 1984; Cumming 1993).

The river bank has always been a focus of human settlement and the condition of riparian areas is a powerful indicator of the catchment quality (Rapport *et al*. 1998). Grazing is the major land use in Australia, occupying approximately 60% of the land surface, and consequently has the biggest impact on riparian areas (Wilson 1990). Because domestic and feral grazing herds concentrate around water sources, riparian and wetland areas suffer to a greater extent than upland areas (Robertson 1997; James *et al.* 1999). Fencing to control stock access to the riparian lands is the most important management action to maintain riparian vegetation. Establishing wide belts of trees and other vegetation along river banks can reduce soil erosion caused by the grazing animals when accessing water (Price & Lovett 2002).

Grazing has a significant impact on the status of native vegetation due to reduction of both structural and floristic diversity (Robertson 1997; Jansen & Robertson 2001, 2005).

Whereas the role of riparian vegetation in controlling channel and bank stability is well documented, little is known about natural control of stream flow by riparian vegetation. However, during high water and floods, riparian vegetation it thought to increases channel roughness. In low order stream systems, riparian trees reduce solar heating of stream water by shading (Brown & Krygier 1970).

2. 4 Impact of riparian land use on water quality

Human intervention on the natural landscape has a great influence on the watershed hydrology. Studies on impact of whole catchment scale land use on water quality provide some knowledge on the general patterns of water quality because all parts of the watershed will not be influenced by the land use and management (Gove *et al.* 2001). But, condition of riparian land use might be more influential than total land use of whole catchment for water quality in river system. Studies conducted in agricultural basin of north-eastern Nebraska, United State of America revealed that stream water chemistry and invertebrate health and diversity were positively related to riparian land use (Whiles *et al*. 2001). Similarly, Chang (2008) found that land use, topographic and soil factors at the 100 metre riparian buffer had more influence on variation of total nitrogen and total phosphorus than the whole catchment land use in the Han River of South Korea. However, research conducted in Southern Ontario Watershed of Canada, Silva & William (2001) reported that catchment scale land use had greater impact on water quality than the 100 metre buffer scale land use. In the Coal River catchment impact of land use at different scales on water quality remains unknown. Due to the nature of over exploitation of the riparian zone, understanding the role of riparian land use on water quality is important. Basnyat *et al*. (1999) reported that riparian land use was more significant in determining stream nutrient concentrations than land use over the whole river basin.

2. 5 Use of GIS on spatial analysis of land use

In the past, several study have focussed on the spatial and temporal changes in the water quality in river basins such as Seine River in France (Meybeck 2002), the Han River in South Korea (Chang 2005) and the Struma River in Bulgaria (Astel *et al.* 2007). These studies showed that degraded water quality downstream of the river was due to the cumulative effect of activities in the upstream areas in the river. It highlights the importance of studying the variation in spatial land use on the river water quality. To quantify the spatial land use data in a watershed aerial photography or satellite imagery is necessary. GIS software can be used to create land use polygons from aerial photography. Aerial photographs taken at different times provide valuable information on the physical characteristics of the landscape, such as land use and vegetation cover (Borrough & McDonnel 1998). Similarly, digital elevation models (DEM) have been widely used to delineate waterways, watershed boundaries, and subcatchments for each water monitoring station (Mooree *et al.* 1991). These spatial analyses help to identify the watershed landscape features, which are used to study the relationship between land use and water quality (Tong & Chen 2002; Chang 2008). GIS not only can integrate and analyse spatial and temporal data to quantify the land use changes, but it can also help to assess the landscape characteristic very quickly and relate these to the adjacent river water quality parameters. Chang (2008) used spatial data and GIS software to study the spatial patterns of water quality in the Han River in South Korea. Several studies have applied statistical models combined with GIS and remotely sensed data to know how watersheds are linked with the spatial variation of water quality in the river. For example**,** Chang & Carlson (2005) used GIS derived land cover data to examine the relationship between land cover and chloride, total organic carbon, and lead concentration in 10 sub-basins of Spring Creek, Pennsylvania, United States of America. Wang & Yin (1997) used GIS to observe possible links between spatial land use data and water quality data in the Great Miami River, United States of America.

3 Description of study area

3. 1 Characteristics of the Coal River catchment

The Coal River catchment in south east Tasmania covers an area of 540 km^2 (Figure 3). The Coal River is 80 km long from its source south east of Tunnack hill at 580 m altitude and ends at the Pittwater estuary to the south of Richmond. It is surrounded by the Prosser River catchments and Little Swanport in the east, the Jordan River catchment in the west, and the Macquarie River catchment in the north (DPIWE 2003b). Only 1.5 km from Lake Tiberias, the source of the Jordon River, the Coal River channel turns southward into a sandstone gorge for approximately 10 km and from Brandy Bottom at the southern end of gorge it flows through alluvial flats to Richmond. Historical records show that stream flow was ephemeral in nature for its entire length being dependent on rainfall. It has two main tributaries, the Native Hut and White Kangaroo Rivulets, which make significant contributions to the flow. Craigbourne Dam in Coal River was built in 1986 with water storage capacity of 12500 ML as part of the South- East irrigation Scheme (Baker 2000). After construction of Craigbourne Dam river flow became continuous with high and low flow depending on the season. Before the dam construction there was higher and continuous winter- springs base flows and low summer– autumn flows was the characteristic of Coal River. But after construction of Craigbourne dam flow was regulated that means high base flows during summer- autumn, reduced base flows during winter- spring and loss of the natural seasonal pattern in the Coal River below the Craigbourne Dam (Davies 2002). The Coal River runs through forest, grassland, agricultural land supplying water for irrigation and contributing to the economy of the region. Human activities have affected the Coal River system in various ways, for instance, through agricultural development, deforestation, urbanisation on river banks, urban drainage, pollution, and sewerage discharge, and flow regulation (dam and channelization).

The major wooded vegetation found in this catchment is dominated by candle bark (*E. rubida*), white gum (*Eucalyptus viminalis*), silver peppermint (*E. tenuiramis*), stringybark

(*E. obliqua*), silver wattle (*Acacia dealbata),* native cherry (*Exocarpos cupressiformis*), pink heath (*Epacris impressa*), and bracken (*Pteridium esculentum*) (Davies 1988; Gallagher 1997). Where human intervention is less apparent, riparian vegetation is dominated by black gum (*Eucalyptus ovata*) and white gum (*Eucalyptus viminalis*) in the alluvial flats (Askey-Doran 1993). In areas cleared for agricultural purposes, the riparian zone and stream banks are heavily infested with exotic species including willow (*Salix fragilis*), gorse (*Ulex europaeus*), blackberries (*Rubus fruticosus*), hawthorn (*Crataegus momgyna*), African boxthorn (*Lycium ferocissimum*), sweet briar (*Rosa rubiginosa*), and Californian thistle (*Cirsium arvense*) (DPIWE 2003b).

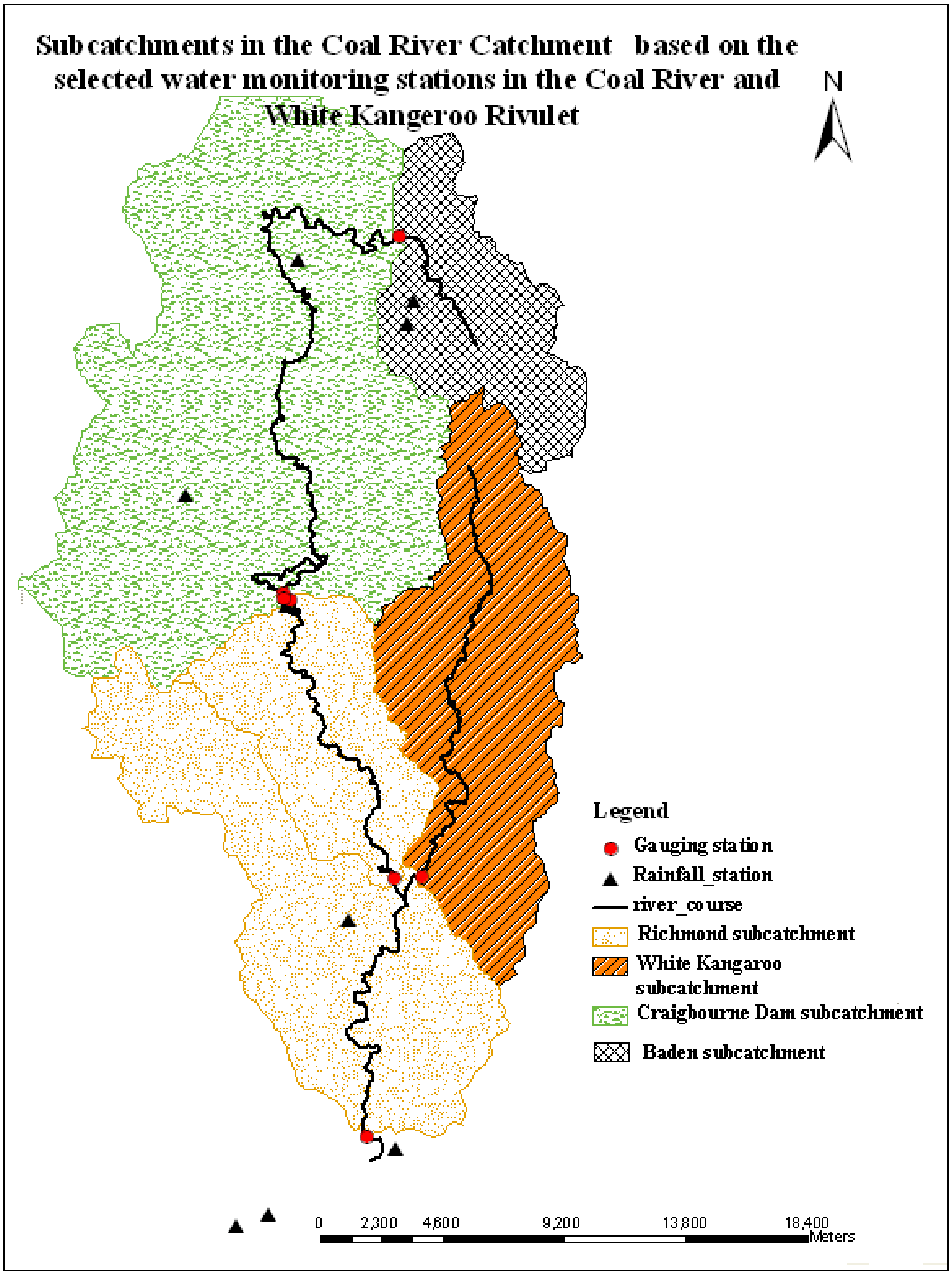


Figure 3 Coal River valley catchment and subcatchments based on the location of gauging stations on the Coal River and its tributaries.

3. 2 Climate

The Coal River catchment lies within one of the driest regions of Tasmania with a mean annual rainfall ranging from 500 to 600 mm across the catchment. The rainfall is highly variable from year to year and month to month and is typically prone to drought conditions. The catchment also has large inter-annual variability in rainfall controlled by the topography with higher rainfall occurring around the upland areas in the north, west and east of the catchment (DPIWE 2003b). The mountains to the west of the catchment place much of the valley within a rain shadow (Gallagher 1997). These variations have a profound effect on natural vegetation and agricultural activities. Average monthly rainfall varies between 18.56 mm and 91.63 mm as shown for Tunnack ("Blue Horizon"), Colebrook ("The Meadows") and Richmond ("Lowland") in figure below.

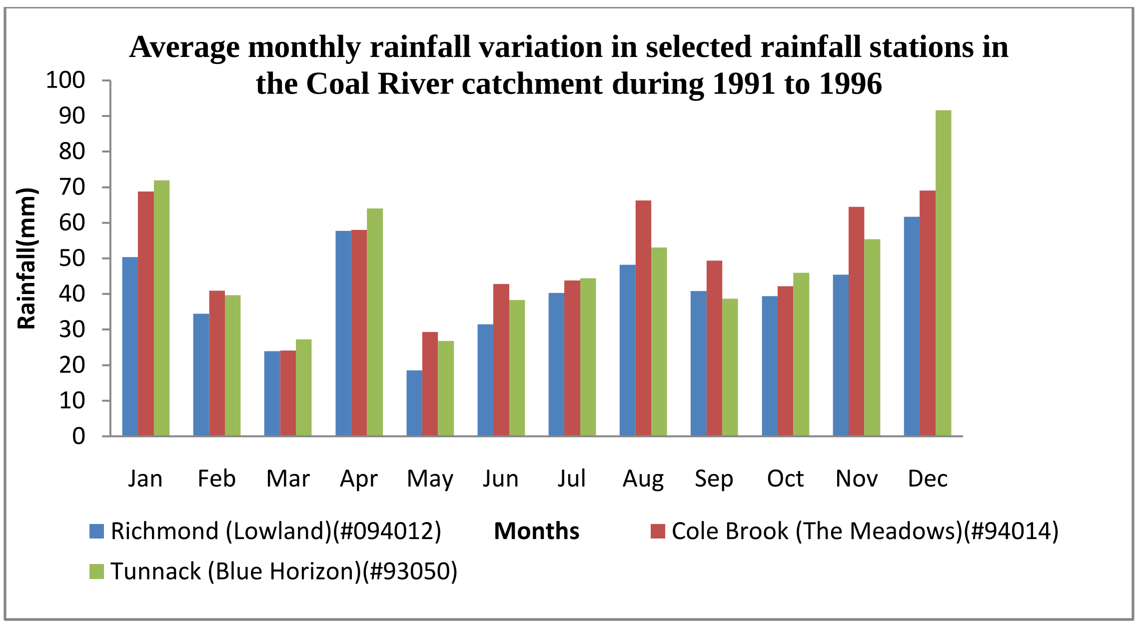


Figure 4 Average monthly rainfall for meteorological stations in the Coal River valley.

3. 3 Topography

Topography, soil and geology have a significant modifying effect on stream flow, sedimentation, soil dispersion and nutrient translocation. Similarly, topography and landforms are greatly affected by the rock type. Both of these factors strongly influence soil erosion potential (Grice 1995) drainage and consequently land use activities. From Campania to the river mouth, the Coal River Valley consists of a flat plain up to 5 km wide. Below the Craigbourne Dam site, the river is restricted within a dolerite gorge making a narrow seep-sided V-shaped valley. Further downstream the valley floor slowly widens into a broader valley in which a series of Quaternary alluvial floodplains and terraces have developed (Leaman 1971).

3. 4 Geology and soil types

Soil types in the Coal River catchment have been influenced by the geological history of the catchment (DPIWE 2003b). Basically the Coal River system is created by extensive Tertiary faulting (90-60 million yrs BP) which shaped the elongate valley. It is surrounded by dolerite capped ridges underneath which lie Triassic sandstones, Permian siltstones and mudstones (Leaman 1971). In the upper reaches the Coal River flows northward across Jurassic dolerite and then for the most part on Triassic sandstones. At Baden, the river circles west, meandering its way through a mixture of siltstone and sandstone.

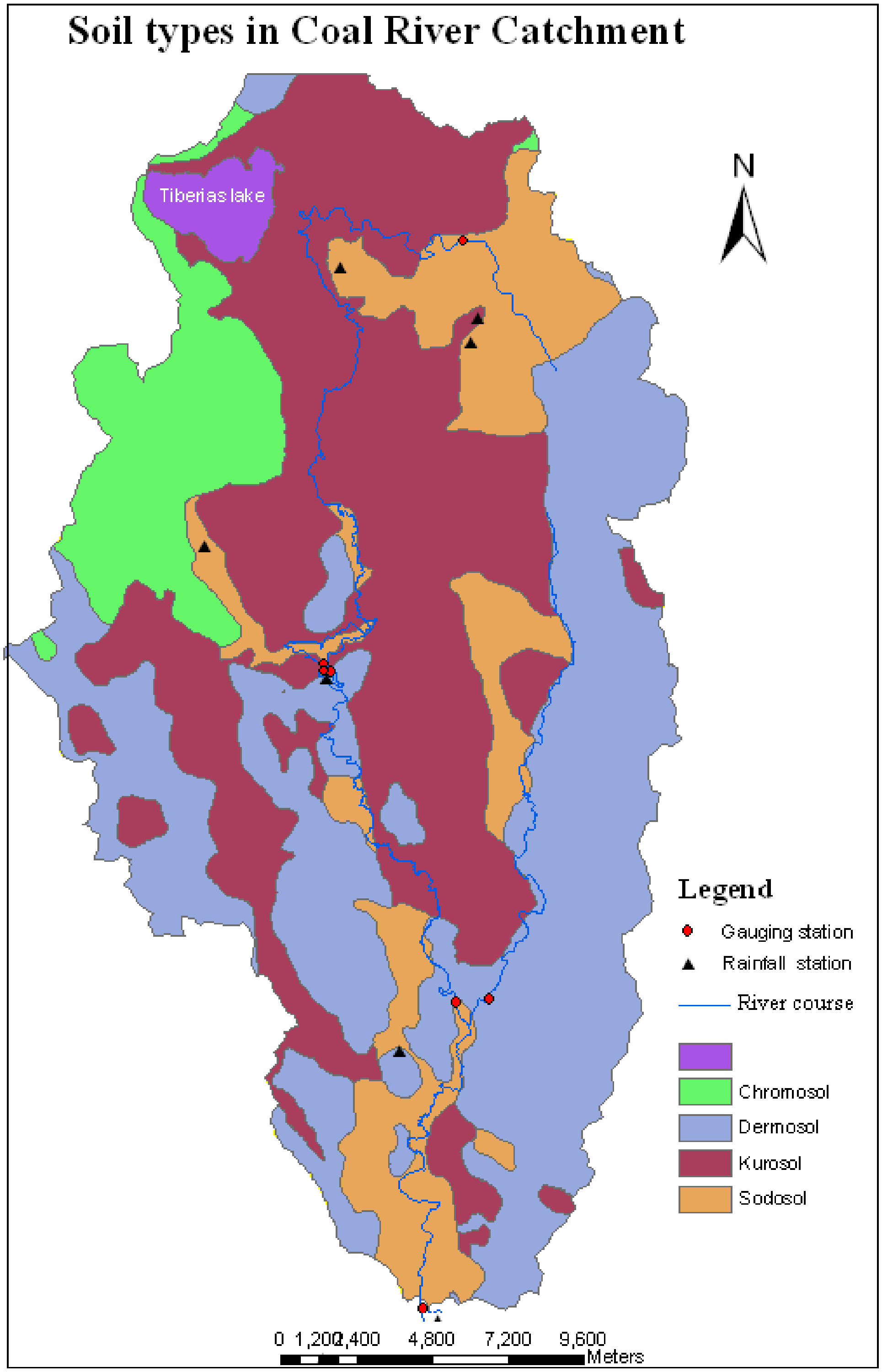


Figure 5 Soil types in Coal River Catchment. (Source: Information and Land Service Division, Department of Primary Industry, Park, Water and Environment, Tasmania).

The effect of climate, topography, flora and fauna acting on parent material over time reveals the spatial distribution and the properties of the soils. Throughout the valley Quaternary alluvial deposits dominate while clayey Tertiary sediments of alluvial and lacustrine origin which have filled in some lower parts of the valley (Leaman 1971; DPIW 2003b). In the Coal River catchment soils develop on a wide range of parent materials including windblown sands, dolerite, mudstone, sandstone and Tertiary sediments. Among the others soil classes, sodic soils are also found in the Coal River valley (Holz 1987; Doyle & Habraken 1993). There are Tertiary sedimentary deposits along the Coal River Estuary and at Pitt Water, while basalt extends between Campania and Richmond on the Coal River plains. The stream valleys throughout the catchment are primarily Quaternary alluvial deposits (Leaman 1971).

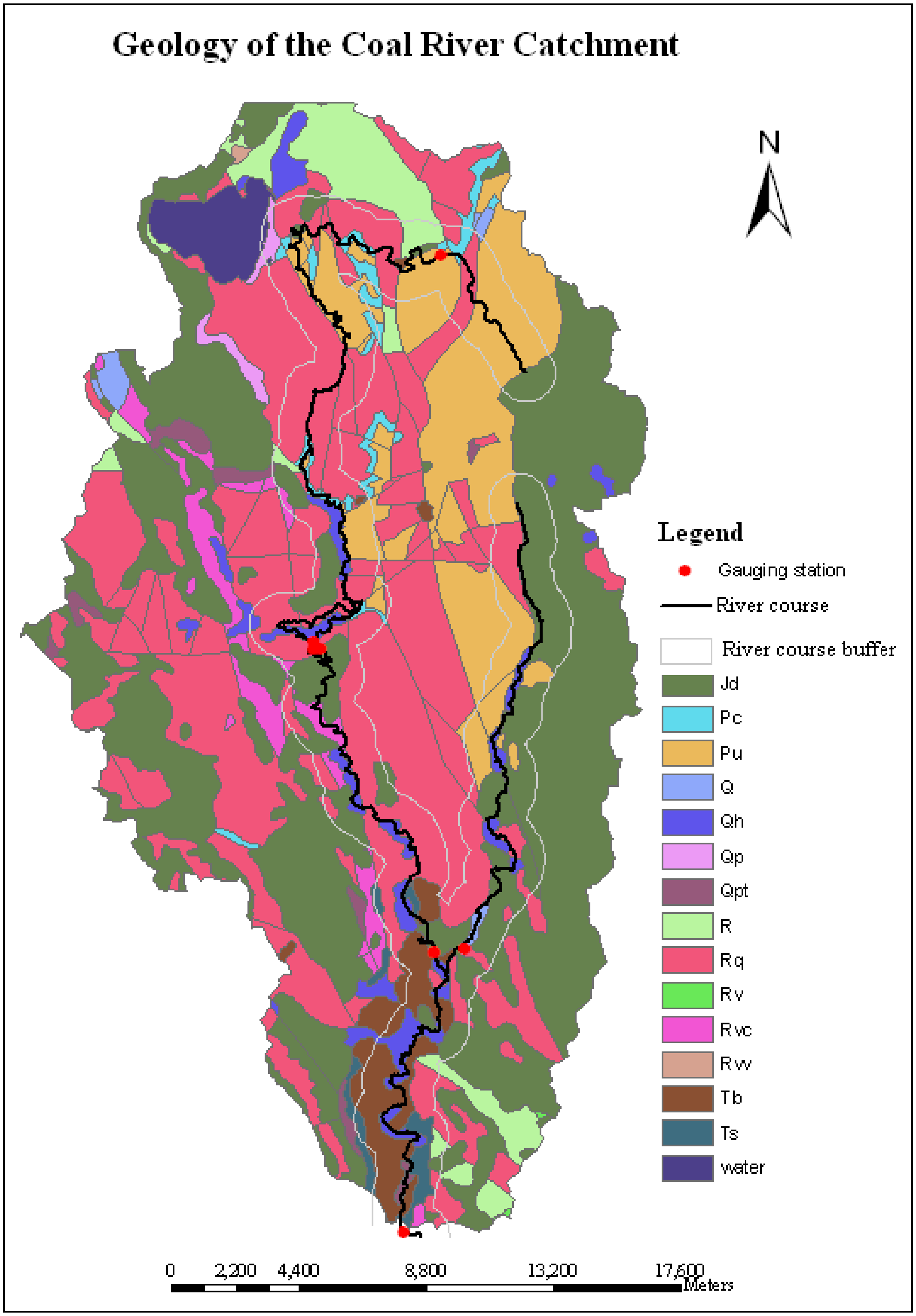


Figure 6 Geology of the Coal River Catchment.

Geological Legend

Q	Quaternary, undifferentiated sediments (mixed-Soil Parent Material [SPM])
Qh	Quaternary sand gravel and mud of alluvial, lacustrine and littoral origin (mixed SPM)
Qp	Quaternary glacial, periglacial and fluvioglacial sediments including till and interglacial deposits (mixed SPM)
Qpt	Quaternary talus, vegetated and active
Tb	Tertiary basalt (tholeiitic to alkali – mafic SPM) and related pyroclastic rocks (mafic SPM)
Ts	Tertiary, dominantly non-marine sequences of gravel, sand, silt, clay and regolith (mixed SPM); water- dam and lakes.
Jd	Jurassic dolerite (mafic SPM) with locally developed granophyre
R	Triassic fluvio-lacustrine sequences of sandstone, siltstone and mudstones (siliceous SPM);
Rq	Dominantly quartz sandstone (highly siliceous SPM)
Rv	Dominantly lithic sandstone with felsic volcani-clastics (siliceous SPM)
Rvc	Lithic sandstone, siltstone and mudstone (siliceous SPM) with some coal (organic) and basal quartz sandstone (highly siliceous SPM)
Rvv	Dominantly siltstone, lithic sandstone, and mudstone (siliceous)
Pc	Permian freshwater sandstone (highly siliceous SPM) with coal measures (organic SPM)
Pu	Permian upper glacio-marine sequences of pebbly mudstone (siliceous), pebbly sandstone (highly siliceous SPM) and limestone (calcareous SPM)

Source: Tasmanian Geological Survey, Geology of Southeast Tasmania).

3. 5 Land use and management practices

Land use patterns in the area have a noticeable impact on environmental management and planning. Urbanisation of rural areas within the catchment at Campania and Richmond, forest clearing for both plantation and agricultural use, increased intensive cropping, and over grazing of pastoral land have all caused environmental impacts in terms of surface runoff and stream quality. The Coal River catchment has a wide diversity of land uses, including improved pastures, native pastures, native and plantation forestry, irrigated cropland, nature conservation areas, recreation, irrigated cropland and rural residential areas (DPIWE 2003b; Figure 8). In the upper part of the Coal River catchment, forestry and grazing are the main land use types while lower parts of the catchment are used extensively for agricultural purposes including grazing (Figure 7).

Figure 7 Land use in Coal River catchment (Intensive grazing on hill slopes with bare soils and cultivation in the fore ground).

Farmers practice crop rotation under dryland farming where pastures are rotated with barley, oats, wheat, poppies. In areas that are not suitable for commercial crops, pastures are rotated with fodder crops such as oats and turnips (Daley 1999). When Craigbourne Dam was built in 1986 traditional farming practices moved away from dryland cropping and grazing to higher value irrigated crops including vineyards, poppies, cherries and vegetables (DPIWE 2003b). The dam can supply approximately 3000 mega litres of water per year which not only increases the agricultural productivity in the valley, it also has the potential to redistribute salt through irrigated agriculture which can lead to increased soil and water salinity.

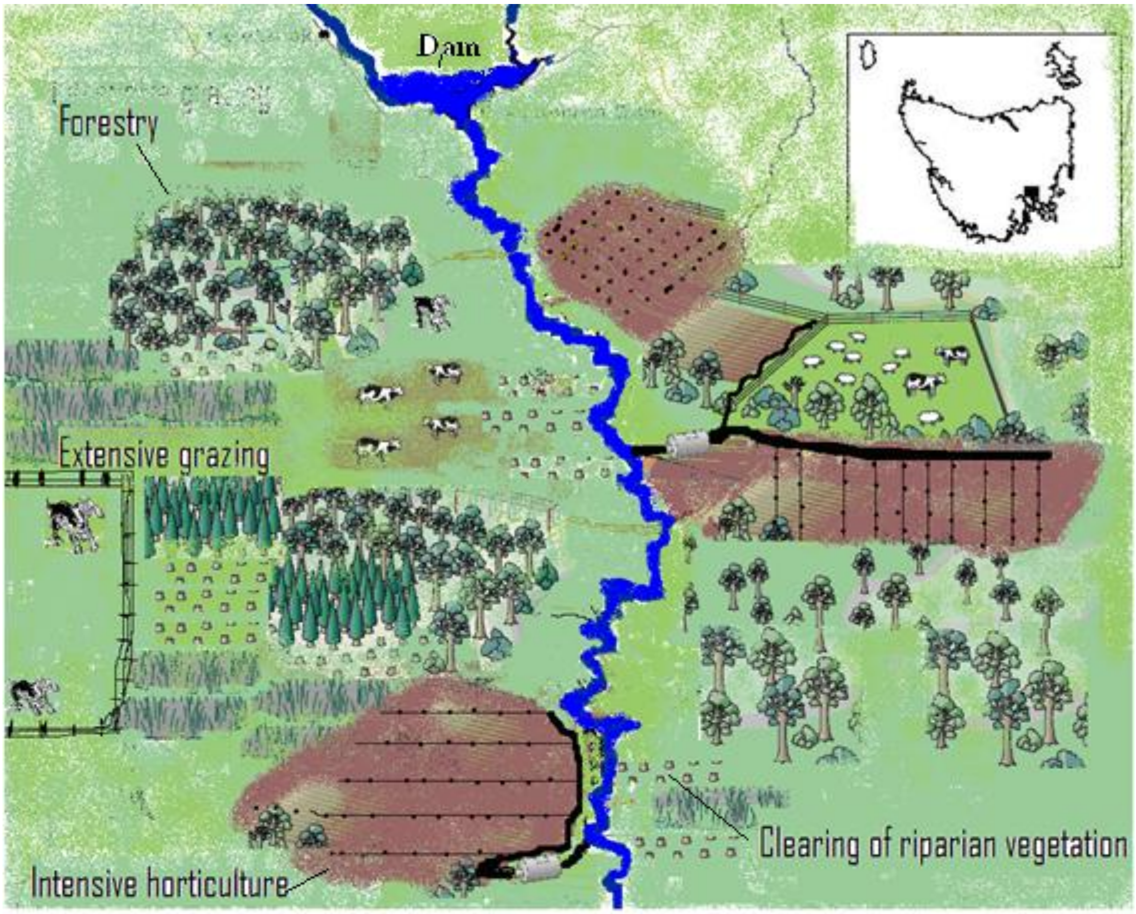


Figure 8 Conceptual diagram of land use in the Coal River valley.

The main indicators of changes in land cover are loss of riparian vegetation and increased sediment loads in rivers. In some part of the river willows and black berries have infested the river channel which has affected the river flow (Figure 9). To maintain a healthy river ecosystem, riparian vegetation can play a number of vital roles. Present farming practices such as maintaining hedges along paddock farm boundaries, clearing forest for grazing land, establishing perennial horticulture (vineyard, cherries and olive) may have an effect on the catchment environment (Figure 10). The focus of the past studies such as Holz (1987), Daley (1999) and Read (1999) has been on soil types, change in land cover, climate and stream flow and comparison of in-stream fauna with willows and other riparian types of the Coal River catchment but not on the effect of land use on water quality.

Figure 9 Willow and black berries infestation in the Coal River completely covering the channel so that the water flow is no longer visible.

It is known that the health of rivers is directly influenced by the condition of their riparian zones. Trees hanging over streams and rivers reduce light penetration into the water and consequently lower the in-stream water temperature which influences aquatic biodiversity. This is the primary reason why large scale willow removal program were initiated in the Coal River in 1990 (Bobbi 1999; Figure 11).

Figure 10 An olive grove in the Coal River valley.

Figure 11 Willow removal in the riparian zone in the middle of the Coal River valley.

3. 6 Water conditions and availability

The intensity of land clearing that has taken place through time is understood to have contributed to elevated salt levels in waterways (DPIPWE 2003a; DPIPWE 2003b). As a result of land clearance and replacement of forest and woodland with shallow rooted crops and pastures, less rainfall is taken up through transpiration which consequently soaks into the ground, fills the shallow aquifers and brings natural salt to the surface. Irrigation practices also contribute to this process. In years of very low rainfall this problem is worsened by highly saline groundwater sources contributing to base flow in waterways. The Coal River catchment has a considerable volume of groundwater (Leaman 1971), so in the drier summer months when the major waterways are prone to low or no flow and flow is maintained from groundwater stores. In the winter-spring months, surface flows are higher and more continuous. Therefore, flood and high flow events in the waterways are highly variable in magnitude and timing, both within and between years. Water extraction rights exist for landowners whose land is on either side of the waterways which provides them with the right to extract a limited quantity of water that can be stored for domestic use or irrigation.

4. Methodology

Two types of data were needed to complete this study, namely water quality data and land use or land cover data.

4. 1 Description of available data

This study examined the relationship between historical water quality data and land use data for the Coal River and White Kangaroo Rivulet. The spatial variation in selected water quality parameters was also studied. For this study, random monthly grab sample water quality data was utilised from the period July 1999 to February 2008 as there were not sufficient continuous time series data from all studied stations during that period. However, short term continuous data were used for spatial and temporal comparison of some parameters. For some variables there were very few observations, consequently only selected variables such as pH, electrical conductivity, dissolved oxygen, turbidity, temperature, flow, nitrate, total nitrogen, dissolved reactive phosphorus, and total phosphors with a high number of observations were used. To compare the effect of land use on water quality, only water quality data and land use data from 2005/7 was used. The land use data were generated from the aerial photograph for years 2005/7.

4. 2 Aerial photograph

Orthorectified aerial photographs were obtained from the Department of Primary Industry, Parks, Water and Environment. It required sixteen 2005/7 photographs to obtain full coverage of the study area in the Coal River Catchment. Due to cost and time constraints only 2005 and 2007 orthorectified aerial photographs were obtained from DPIPWE to generate the land use polygons for the study area. A descriptive summary of the aerial photographs used for this project is given in Table 2.

Table 2 Summary of aerial photographs used in study.

Year	Title	Project	Flown by	Film	Negative	Run	Flown	Height	Lens	Scale	Type
2005	Central east revision	A118	Aerotech	1396	85	31	06-Nov-05	22500'	153	1:42000	Color
2005	Central east revision	A118	Aerotech	1396	87	31	06-Nov-05	22500'	153	1:42000	Color
2005	Central east revision	A118	Aerotech	1396	133	30	06-Nov-05	22500'	153	1:42000	Color
2005	Central east revision	A118	Aerotech	1396	136	30	06-Nov-05	22500'	153	1:42000	Color
2005	Central east revision	A118	Aerotech	1396	138	30	06-Nov-05	22500'	153	1:42000	Color
2005	Central east revision	A118	Aerotech	1396	163	29	06-Nov-05	22500'	153	1:42000	Color
2005	Central east revision	A118	Aerotech	1396	165	29	06-Nov-05	22500'	153	1:42000	Color
2007	Central east revision	A110	Aerotech	1421	198	32	13-Feb-07	22500'	153	1:42000	X100
2007	Central east revision	A110	Aerotech	1421	200	32	13-Feb-07	22500'	153	1:42000	X100
2007	Central east revision	A110	Aerotech	1421	202	32	13-Feb-07	22500'	153	1:42000	X100
2007	Central east revision	A110	Aerotech	1421	234	33	13-Feb-07	22500'	153	1:42000	X100
2007	Central east revision	A110	Aerotech	1421	236	33	13-Feb-07	22500'	153	1:42000	X100
2007	Central east revision	A110	Aerotech	1421	258	33	13-Feb-07	22500'	153	1:42000	X100
2007	Central east revision	A110	Aerotech	1421	259	33	13-Feb-07	22500'	153	1:42000	X100
2007	Central east revision	A110	Aerotech	1421	260	34	13-Feb-07	22500'	153	1:42000	X100
2007	Central east revision	A110	Aerotech	1422	16	35	13-Feb-07	22500'	153	1:42000	X100

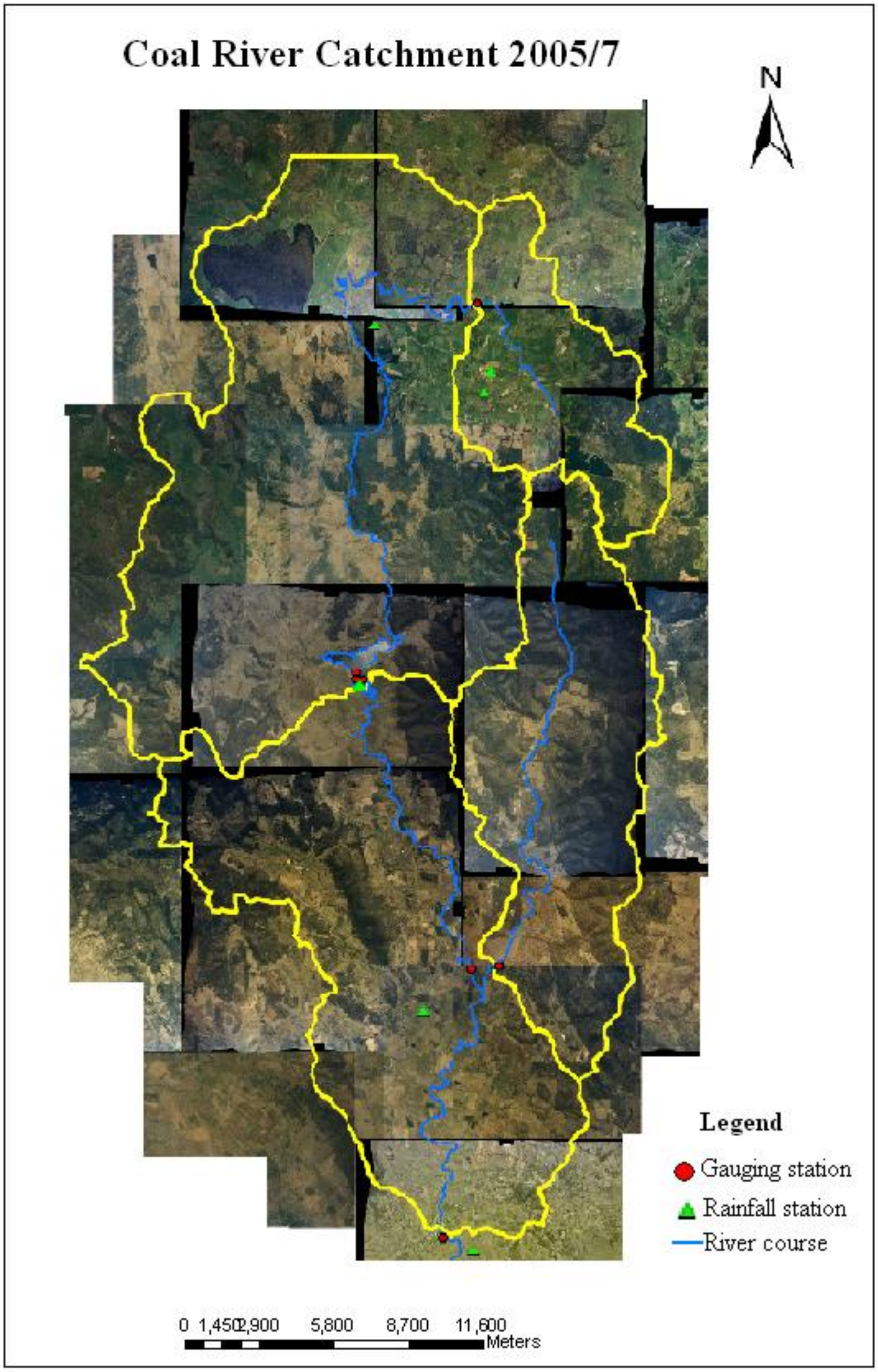


Figure 12 Orthorectified and joined aerial photographs for 2005/7 (catchment and subcatchment boundaries shown by yellow line).

4. 3 Water quality data

All the water quality information used in this study was collected and prepared by the Department of Primary Industry, Parks, Water and Environment (DPIPWE) as part of their stream chemistry monitoring network. In this study four water monitoring stations were chosen. As the periods of available data varied between stations, data for the common period from July 1999 to February 2008 was used in this study. The selected water quality variables and gauging stations are presented in Table 3 and 4.

Table 3 Selected water quality parameters used for study.

Variable	**Units**
Water temperature	^{0} C
Water pH	
Dissolved oxygen	mg/L
Nitrogen as nitrate(NO_3^-)	mg/L
Total nitrogen	mg/L
Dissolved reactive phosphorus	mg/L
Total phosphorus	mg/L
Turbidity	NTU
Electrical conductivity	µS/cm
Flow	Cumecs

Table 4 Data collection station details.

Name	Coal River at Baden	Coal River Downstream Craigbourne Dam	Coal River at Richmond	White Kangaroo Rivulet
ID	3203	3206	3208	3209
Status	Open	Open	Open	Open
Northing	5302124.754	5288480.019	5268280.823	5278134.024
Easting	537324.16	533255.567	536167.098	538206.366
Telemetered	NO	Yes	Yes	Yes
Elevation above sea level (meters)	430	-	-	55
Field measurement data	1992-2009	1992-2008	1990-2008	1992-2008
Nutrient data	2000-2008	1999-2008	1992-2008	1992-2008
Stream flow data	1971-2008	1986-2008	1995-2008	1990-2008

4. 4 Rainfall data

Rainfall data was used from different stations within the catchment from Baden to Richmond (Table 5). All the rainfall data were obtained from the Australian Government, Bureau of Meteorology.

Table 5 Details of rainfall stations.

Station	Station no	Latitude	Longitude	Easting	Northing	From	Height of station above mean sea level(m)
Tunnack Blue Horizon	93050	-42.4403	147.4083	533580.35	5301255.14	1990-2005	420
Tunnack Fire station	94195	-42.4544	147.4614	537939.02	5299667.14	1997-2009	462
Tunnack Post Office	94067	-42.4617	147.4583	537679.74	5298857.94	1922-1990	460
Richmond (Lowland)	94012	-42.7406	147.4553	537266.51	5267889.98	1920-2008	11
Campania (The Pines)	94009	-42.66	147.4325	535444.26	5276483.40	1920-1998	55
Below Craigbourne Dam(Coal River)	94182	-42.5572	147.4033	533107.40	5288276.67	1991-2009	150
Colebrook (The Meadows)	94014	-42.5206	147.3567	529299.05	5292357.88	1911-2009	225

Source: www. bom.gov.au/climate/data

4. 5 Land use data

For this study 2005/7 aerial photographs were used to digitise land use cover. Historical land use data for the study area was not available. So, land use polygons were subsequently generated from 2005/7 aerial photograph using GIS software. To see whether there was significant impact of riparian land use on water quality, the riparian zone was defined in this study as that area within one kilometre either side of the Coal River and White Kangaroo Rivulet. The land use classes used were based on the following characteristics observed in the aerial photographs.

4.5.1 Cereal cropping

Aerial images show clear visual signs of cultivation such as tillage lines, vibrant green colour, linear growth patterns and in some cases irrigation equipment. This land use class also includes poppy cropping. Land classed as cropping at the time the photograph was acquired may in another year be in pasture production or other phase of a rotation.

Plate 1 Aerial photograph showing cereal cropping at Campania.

Plate 2 Ground truthing an area identified as cereal cropping at Campania in January 2010.

4.5.2 Intensive horticulture

Areas cultivated for annual horticultural crops such as lettuce, cabbage and other vegetables were identifiable due to the presence of intensive irrigation facilities. This class was defined as crop plants living for less than two years that are intensively cultivated in well tilled lines with evidence of irrigation lines and weed control facilities.

Plate 3 Aerial photo showing intensive horticulture.

Plate 4 Intensive lettuce production 2 km east of Richmond in January 2010.

4.5.3 Perennial horticulture

Areas of land where fruit, nut, olive trees and vineyards are present. Aerial imagery shows linear trees/vines with windbreaks and irrigation infrastructure such as dams and pump houses.

Plate 5 Aerial photo showing perennial horticulture.

Plate 6 Perennial horticulture at Richmond.

4.5.4 Improved pasture

Cleared, flat to gently undulated land with a uniform vibrant tone on images (green to tan hues) and in some case cut hay bales or cut hay drying in the field.

Plate 7 Aerial photo showing improved pasture at Baden.

Plate 8 Improved pasture in 2010 near confluence of White Kangaroo and Coal rivers.

4.5.5 Native pasture

Uneven land surface with variable tone commonly on steeper slopes where natural grasses are grown with no evidence of tillage.

Plate 9 Aerial photo showing nattive pastures.

Plate 10 Photo showing native pasture above Craigbourne Dam in 2010.

4.5.6 Native pasture + trees/shrubs

Native pastures with isolated paddock trees, shrubs or small remnant blocks.

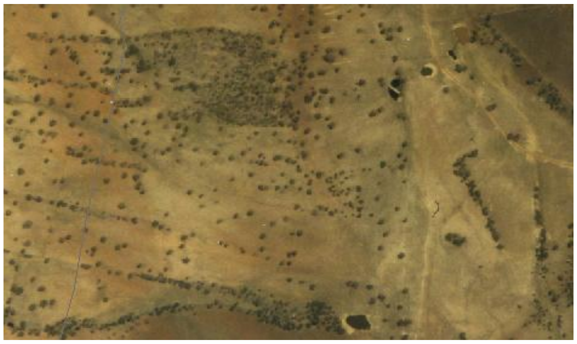


Plate 11 Aerial photo showing mixture of pasture and trees.

Plate 12 Paddock with pasture and trees near Richmond township in 2010.

4.5.7 Dams and Lakes

Areas of dammed or ponded water bodies.

Plate 13 Aerial photo showing Craigbourne Dam.

Plate 14 Photograph showing farm dam at Richmond in 2010.

4.5.8 Native forest

Area includes thick native forest trees such as *Eucalyptus spp.*, sheoaks (*Allocasuarina spp.*) and wattles (*Acacia spp.*)

Plate 15 Aerial photograph showing native forest.

Plate 16 Photograph showing native forest in 2010 near Lake Tiberius.

4.5.9 Plantation forest

Areas where trees species are planted for commercial harvest as indicated by cultivation lines and even stands of single species.

Plate 17 Aerial photograph showing plantation forest in the Baden area.

Plate 18 Photo showing plantation forest in the Baden area 2010.

4.5.10 Home paddocks

Areas of pasture, fruit trees and/or windbreaks with farm sheds and homesteads etc.

Plate 19 Aerial photo showing home paddock.

Plate 20 Photo showing home paddock in 2010.

4.5.11 Willow + riparian trees

Areas of willows and endemic trees species in the river and on adjacent banks

Plate 21 Aerial photograph showing willows and endemic riparian trees on the Coal River.

Plate 22 Photograph showing willows and endemic riparian trees at White Kangaroo station 2010.

4.5.12 Residential areas:

Areas with houses, buildings and others urban infrastructure

Plate 23 Aerial photograph showing residential area of the Richmond.

Summary list of land use types:

- Cereal cropping
- Intensive horticulture
- Perennial horticulture
- Improved pasture
- Native pasture
- Native pasture + trees/shrubs
- Dams and lakes
- Native forest
- Plantation forest
- Home paddock
- Willow + riparian trees
- Residential areas

4. 6 Use of Geographic Information Systems (GIS)

ArcMap version 9.3 GIS Software was used to generate riparian land use polygons in the Coal River catchment from the aerial photography. A digital elevation model (DEM) with a resolution of 25 m was used to derive the hydrological characteristic of the area.

The DEM was obtained from the DPIPWE, Tasmania. Water flows across a surface are influenced by the shape of the surface or the surface morphology. The patterns of surface flow can be examined and predicted through hydrological analysis function within ArcGIS by using surface models. The DEM was used to delineate a drainage system and quantify particular features of that system such as total catchment area. GIS based hydrological models allow for the determination of flow direction, flow accumulation, watershed delineation, and stream network creation (Jenson & Domingue 1988).

A one kilometre buffer zone was delineated using the ArcGIS buffer analysis tool. Spatial analysis tools were used to calculate the areas of each land use types within one kilometre from the Coal River and White Kangaroo Rivulet. A watershed or catchment can be defined as an area that drains water to a common channel as a concentrated drainage. These areas can be derived using the watershed tool in ArcGIS. A pour point is the lowest point of the catchment from where water flows out (Figure 13). In the study area, the pour points are the water monitoring stations. Using the watershed tools, areas that drain into each pour point were delineated by overlaying the water gauging station layer on the catchment and stream network layer. The procedures for delineation of the subcatchment or subwatershed are shown in Figure 14.

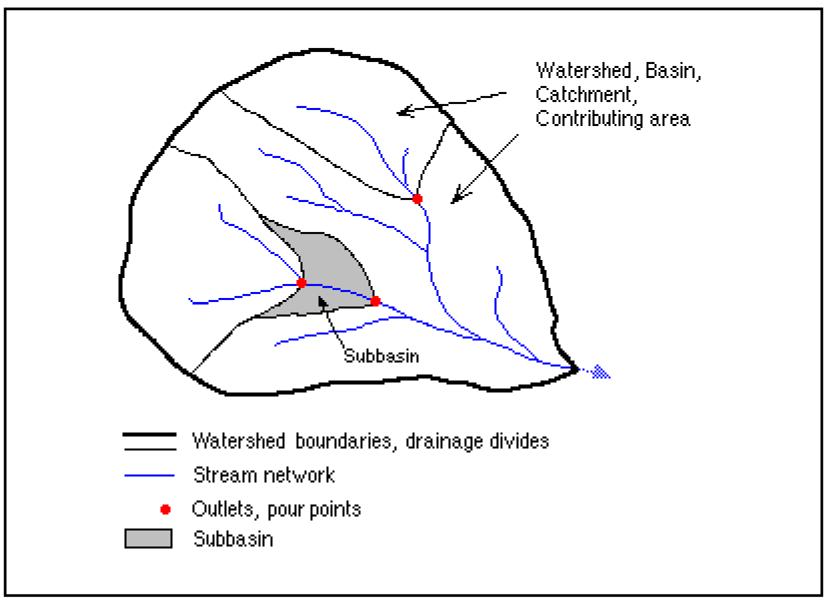


Figure 13 Conceptual diagram of a watershed (source: www.esri.com).

Area and percentage of land use of the riparian buffer were calculated for each subcatchment. The spatial variations of land use and water quality were compared. The means of selected water quality variables from each station and the land use distribution within corresponding subcatchment were analysed using the regression analysis to find out relationships between them.

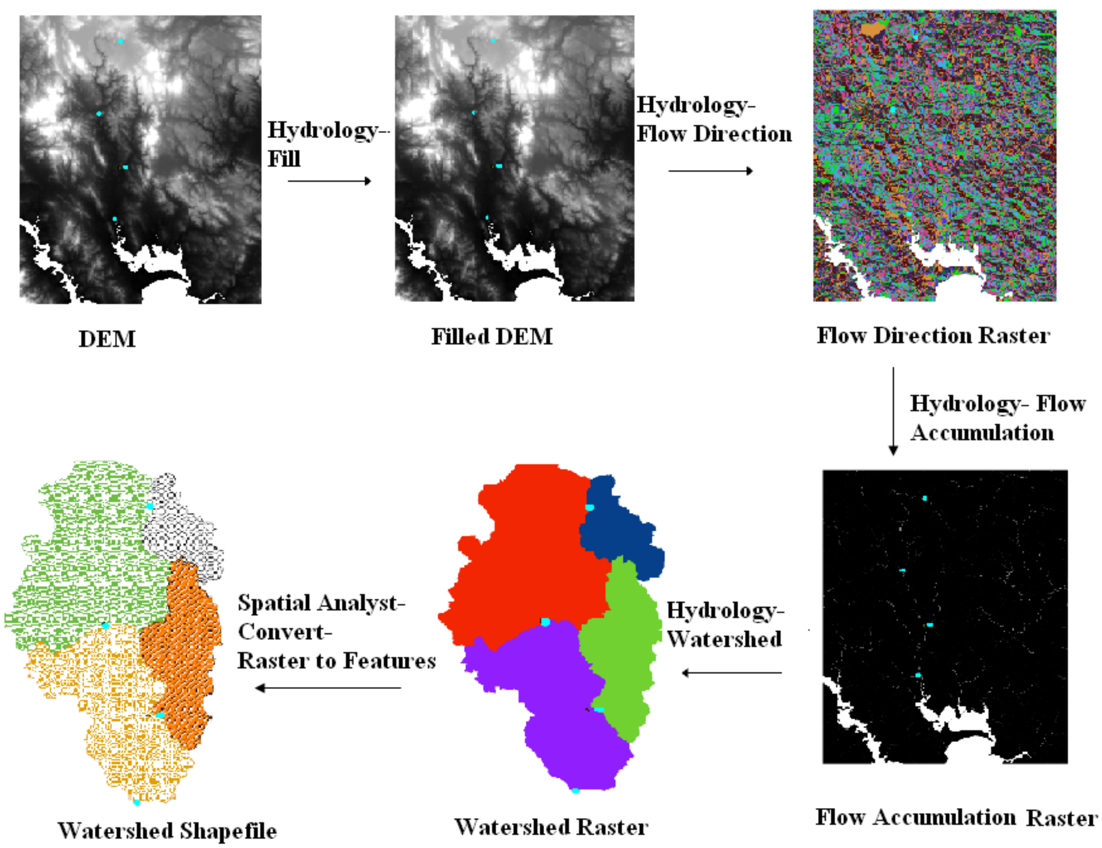


Figure 14 Procedure of subwatershed/catchment delineation.

Temporal analysis of land use and water quality change did not work due to a lack of time series land use and water quality data. So the alternative was to investigate the spatial variation of land use in the subcatchments and its relationships with water quality.

4. 7 Statistical analysis

Spatial variation was examined by comparing the upstream and downstream river monitoring water quality data. Temporal variation and trend is best seen by using continuous data, however in this study we had grab sample data available only for most parameters. Continuous data sets were only collected at Richmond for turbidity and flow so it was not possible to compare the seasonal and temporal variation for all water quality parameters spatially.

Different statistical tests were then applied to water quality data including one way ANOVA, correlation and regression. All the statistical tests were performed using Microsoft Office Excel 2007 and SPSS 17.0 for Windows. The Kolmogorov and Smirnov method was used to test the normality of distribution and where datasets failed this test, square root and natural log transformations were performed as suggested by Townend (2002). Graphs were presented from untransformed data.

In order to test the difference between the gauging station data, a one way ANOVA was performed. Unplanned post hoc tests were used to identify significant differences in the means for the different gauging station data (Field 2000). Levene's test statistics was used prior to means comparisons to find out whether within-group variance differed significantly ($p<0.05$). For data sets with heterogeneous variances and that could not be removed via data transformation, Games-Howell method was used to compare means (Day & Quinn 1989; Field 2005). Due to the heterogeneous variance robust tests of equality of means Welch F- ratio was reported instead of the traditional ANOVA (Field 2005). Pearson coefficients and linear regressions were used to examine the relationships between the water quality parameters. Regression analysis was used to identify casual relationships between dependant and independent variables by fitting a straight line to a set of observations. Stream flow and rainfall data were used as independent variables against the dependant water quality variables to find if the significant relationship could be recognized.

Water chemistry data from different gauging stations were analysed and presented in the form of box-plots (or box and whisker plot) to provide a visual impression and distribution of variability within stations. Boxes with large spread indicate a high level of variation in the data. These plots were also used to identify differences between the stations. They show the maximum, minimum and median recorded values with the bottom and top of the box show the location of first (Q1) and third (Q3) quartiles, the whiskers represent the lowest and highest observation points, the line dividing box representing the median value with the extreme values represented by asterisks. In general, the water quality data is not presented in the form of means as the data is not normally distributed (Dates 1998). Given the skewed nature of water quality data, medians are reported as the best nonparametric statistics for comparison (Christian *et al*. 1991).

5. Results and Discussion

5. 1 Spatial variations in water quality parameters

5. 1. 1 Temperature

Water temperature data was recorded at Baden, downstream of Craigbourne Dam, Richmond and White Kangaroo Rivulet from 1999 to 2008. Figure 15 shows a bar chart of the temperature data. Mean temperatures indicate there is no significant difference between the sites. The error bars indicate the 95% confidence interval around the mean while the top of the bar shows the observation mean. Using 95% confidence intervals makes it difficult to identify significant difference in the means between the groups (Payton *et al.* 2003). So, to detect the statistical difference a post hoc test was performed. A one way ANOVA was used to test the difference in water temperature between the four stations. There was no statistically significant variation in temperatures from the four sites at $p<0.05$ (Appendix 1.4). However, the highest value (27 ºC) was recorded in the Richmond and the lowest (3.3 ºC) in White Kangaroo Rivulet. The box plots (Figure 16) show that water temperatures were highest downstream of Craigbourne Dam and Richmond with median values of 13.40 ºC followed by the White Kangaroo Rivulet station (11.9 ºC) and Baden (10.5 ºC).

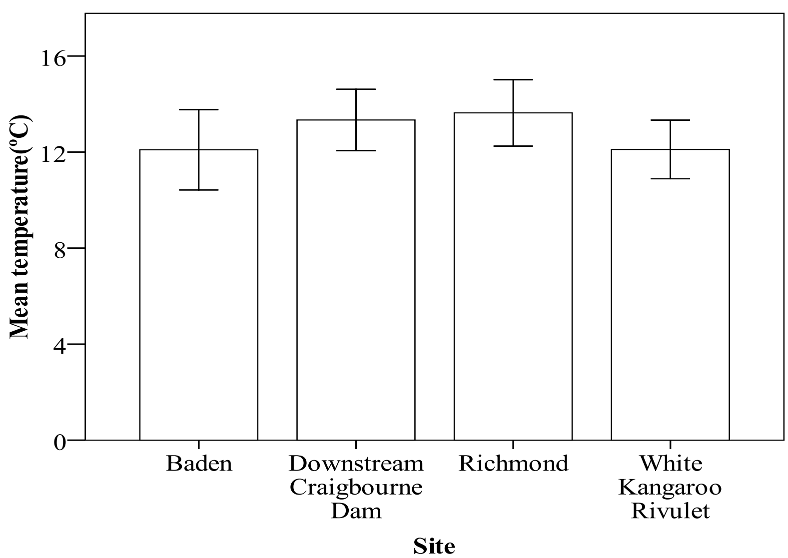


Figure 15 Mean water temperature recorded at four monitoring stations between 1999 and 2008. Error bars indicate 95% confidence interval of mean.

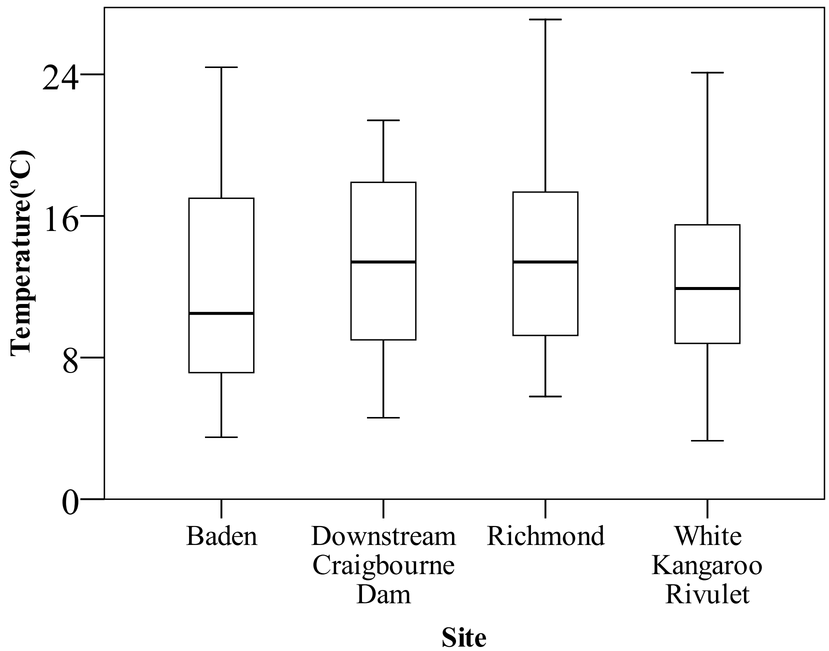


Figure 16 Variability in water quality data recorded at four monitoring stations between 1999 and 2008.

Continuous data collected in September 2004 shows daily variation in the water temperature at three different gauging stations in the Coal River (Figure 17)**.** In this case the Coal River at Richmond shows the highest daily temperature suggesting the influence of a wider channel and subsequent increased sunlight.

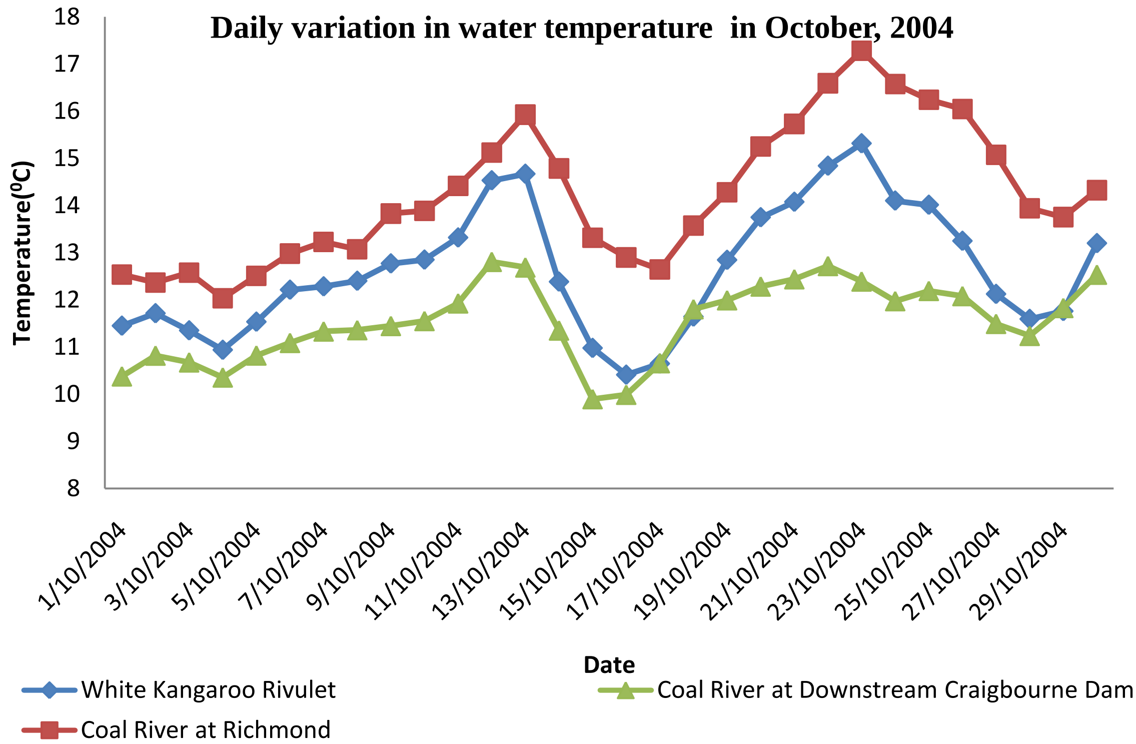


Figure 17 Daily variation of the water temperature in different water monitoring stations.

Monthly temperature from the same stations shows temperatures of 6 - 8 ºC in winter at all the monitoring sites and temperatures of 16 -19 ºC in summer season (Figure 18) with temperatures again higher at Richmond year round.

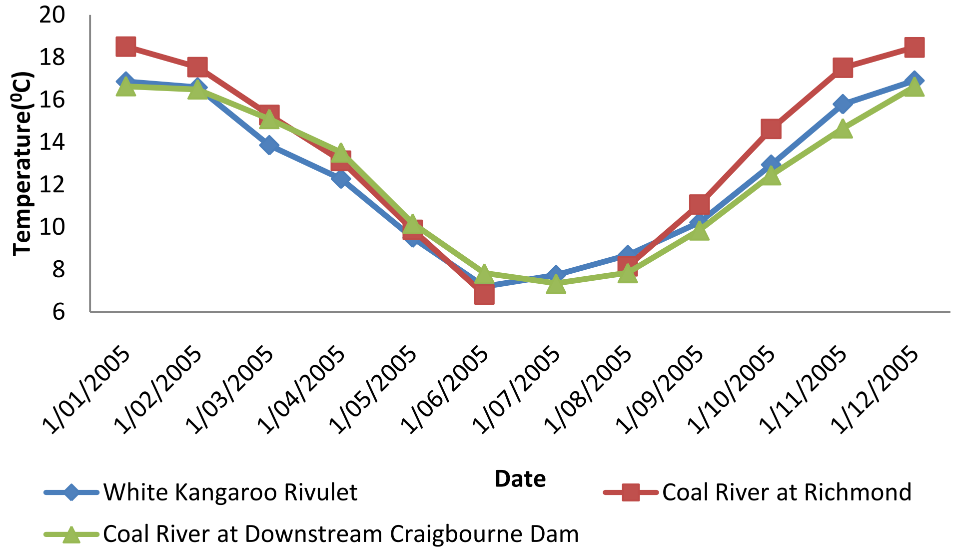


Figure 18 Monthly variation of water temperature in the Coal River in 2005.

The lower temperatures at Baden may be the result of high altitude while at White Kangaroo Rivulet may be due to native forest cover. High water temperatures at recorded Richmond station could be the effects of urban environment resulting in thermal pollution as well as the wider channel and lower flows. Urbanisation, increased area of impervious surfaces and accumulation and drainage of rainwater from roads and paved areas help to increase the stream temperature (Pluhowski 1970).

5. 1. 2 Dissolved oxygen

The Figure 19 shows dissolved oxygen data for the four stations. The observation mean shows that downstream of Craigbourne Dam was significantly higher than the other three stations. Welch F-ratio showed there was a significant spatial effect on the concentration of dissolved oxygen recorded, F (3, 111.427) = 22.19, $p<0.001$. Games-Howell post hoc comparisons of the four stations indicate that downstream of Craigbourne Dam (M = 10.47, 95% CL [10.12, 10.83]) showed a significantly higher mean concentration of dissolved oxygen than Baden (M = 8.08, 95% CL [7.43, 8.73]), Richmond (M = 8.84, 95% CL [8.33, 9.36]) and White Kangaroo Rivulet (M = 8.22, 95% CL [7.46, 8.98]), $p<0.001$.

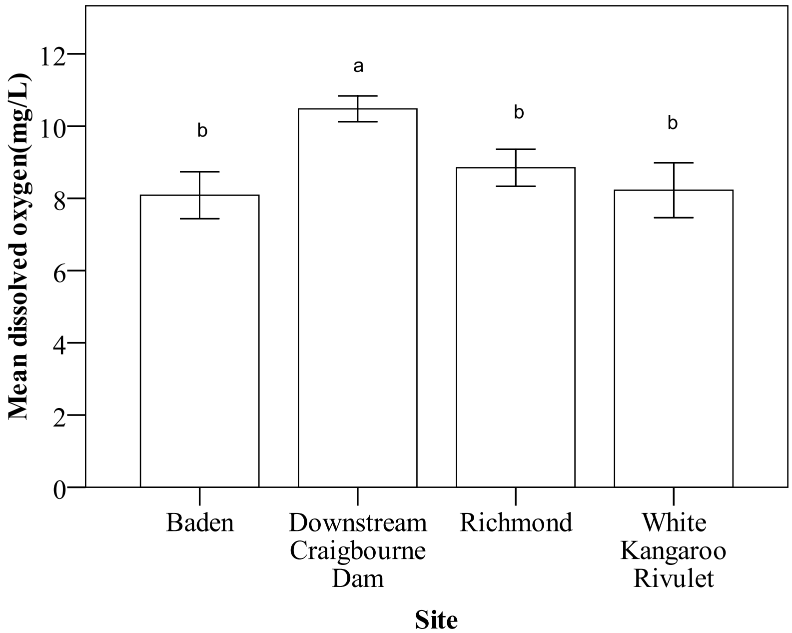


Figure 19 Mean dissolved oxygen data collected at four monitoring stations between 1999 and 2008. Error bars indicate 95% confidence interval of mean. Lower case letters denote results of Games-Howell test ($p<0.05$), used for a post hoc comparisons between means.

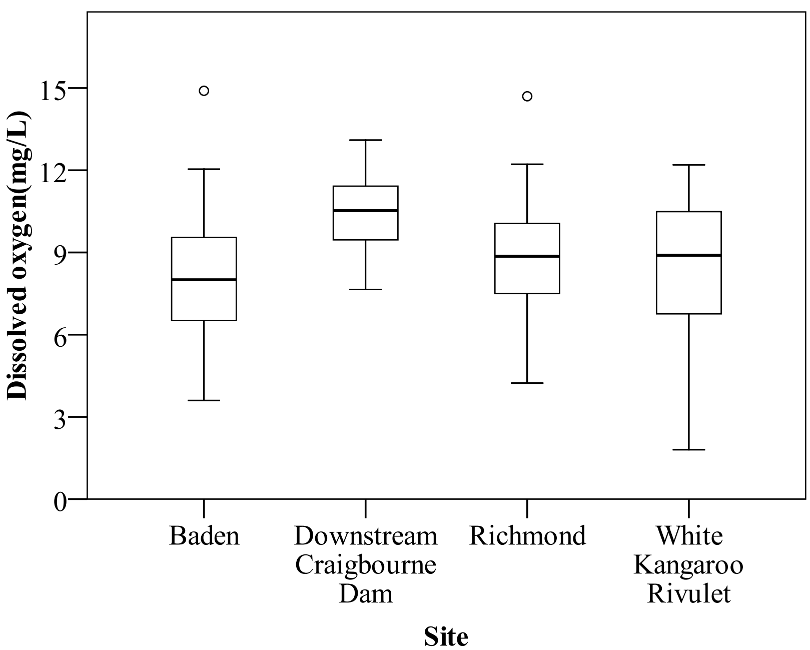


Figure 20 Variation in dissolved oxygen concentrations at four monitoring stations between 1999 and 2008.

The box and whisker plot shows the maximum, minimum and median recorded values. The median dissolved oxygen in downstream of Craigbourne Dam was higher (10.53 mg/L) than Baden (8.00 mg/L), Richmond (8.86 mg/L) and White Kangaroo Rivulet (8.90 mg/L) (Figure 20). The lowest DO concentration (1.8 mg/L) was recorded at White Kangaroo Rivulet in March 8, 2001.

The level of dissolved oxygen (DO) in the stream is very important because of its significant impact on the aquatic flora and fauna. According to ANZECC (1992) guidelines, when dissolved oxygen is below 6 mg/L it is considered a health hazard to aquatic animals and if possible measurements should be taken during the night time to identify the lowest values. In this study all data was recorded during the day. The data indicates there was less variability in the concentration of dissolved oxygen downstream of the Craigbourne Dam which could be the effect of photosynthetic activities in the dam. The higher oxygen levels measured below the dam may in part be due to turbulent release and mixing as the water flows out from the release valve. Bobbi (1997b) reported that maximum DO concentration was 13.5 mg/L in downstream of the Craigbourne Dam and suggested water is oxygenated when released from nose-cone of the outlet valve of the dam. As the nutrient data shows higher concentrations in the dam, this suggests the development of photosynthetic algae. So, dams with higher level of photosynthetic activities tended to have higher levels of oxygen production. However, these values are higher than the ANZECC guideline of dissolved oxygen for aquatic health but less than the site specific trigger values mentioned in the DPIPWE Baseline Water Quality Monitoring Program (DPIW 2008). Dissolved oxygen concentration is usually lower at night and morning and rises to a maximum in the afternoon. The values used in this study were taken during the day time and are still within the acceptable range of trigger values for aquatic life (McNeely *et al.* 1979). Bagalwa (2006) reported that low dissolved oxygen concentration was due to increased organic matter accumulation that triggered an increased decomposition rates by bacteria consuming oxygen in the water.

5. 1. 3 Turbidity

Turbidity at the station downstream of Craigbourne Dam was significantly lower than that at Baden but not significantly different to Richmond or White Kangaroo Rivulet stations (Figure 21). Welch F-ratio showed there was significant variation in mean turbidity between the stations $F(3, 96.28) = 5.07$, $p = 0.003$. The Games-Howell post hoc comparison reveals that downstream of Craigbourne Dam (M = 3.42, 95% CI [2.83, 4.01]), had significantly lower mean turbidity than the Baden (M = 5.95, 95% CI [4.28, 7.61]), Richmond (M = 5.75, 95% CI [2.17, 9.33]) and White Kangaroo Rivulet (M = 7.69, 95% CI [4.49, 10.90]), $p = 0.003$. There was no significant difference between the rest of the groups at $p<0.05$. The box plot (Figure 22) shows the distribution of turbidity data at the four sites.

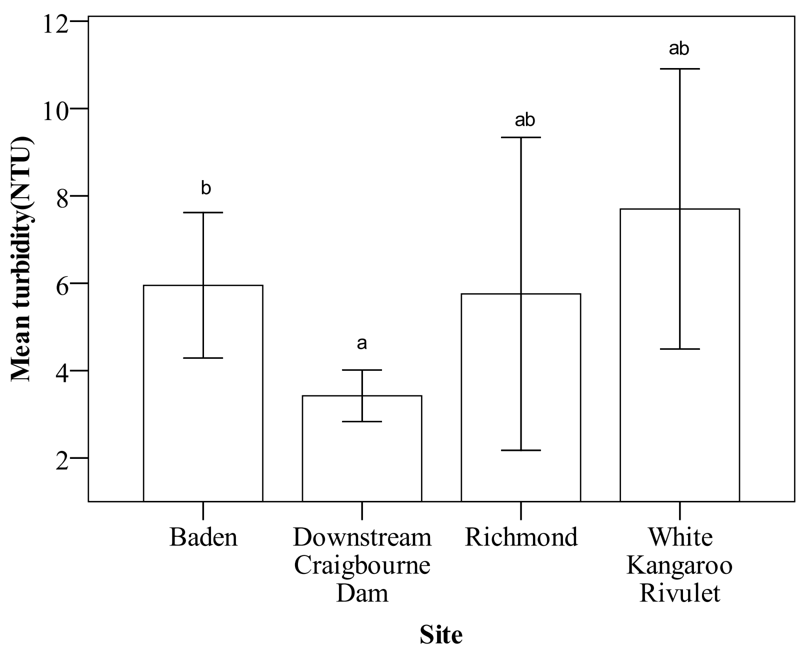


Figure 21 Mean turbidity data collected at four monitoring stations between 1999 and 2008. Error bars indicate 95% confidence interval of mean. Lower case letters denote results of Games-Howell test ($p<0.05$), used for a post hoc comparisons between means.

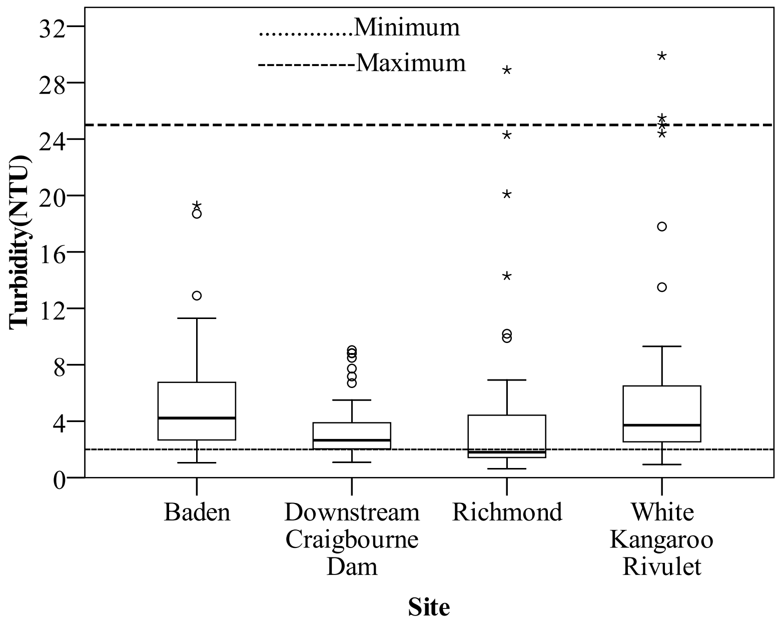


Figure 22 Variation in turbidity data at four monitoring sites from 1999 to 2008 with the dotted lines representing the ANZECC 2000 default low risk trigger values for upland rivers. DPIW 2008 site specific trigger values (80^{th} percentile) are Baden 9 NTU, Downstream Craigbourne Dam 3 NTU, Richmond 6 NTU and White Kangaroo Rivulet 7 NTU.

Turbidity is a measurement of the capacity of light to penetrate water and the value generally represents the amount of suspended materials in the water. Higher values indicate high level of suspended materials in the water. Significant spatial variation in turbidity in the Coal River was observed only at the station downstream of Craigbourne Dam although the highest value (93.1 NTU) was recorded at the Richmond gauging station in August 10, 2004. The higher value for this date may be due to the moderate rainfall (20 mm) throughout the catchment. The lower value recorded downstream Craigbourne Dam is likely to be due to sediment being trapped in the dam before water is released to the measuring station. The higher mean turbidity of the data from White Kangaroo Rivulet may be caused by human activities such as farming and forestry operations.

5. 1. 4 Electrical Conductivity

Electrical conductivity data (Figure 23) shows that salinity at Richmond and White Kangaroo Rivulet are high but not significantly different at 859 and 839 μS/cm respectively, within their site specific low risk trigger values (Richmond 651-981; White Kangaroo Rivulet 402-1146 μS/cm) but nearly three times higher than the ANZECC 2000 default low risk trigger value of 125 for lowland rivers (DPIWE 2008). Salinity at Baden, and downstream of Craigbourne Dam are lower and significantly different at 564 and 441 μS/cm respectively and also within their site specific trigger values. Welch F-ratio, $F(3, 105.70) = 41.09$, $p<0.001$ shows there were significant difference on the mean recorded on different stations. Similarly to find out where the differences among four stations occur, Games-Howell post hoc comparisons were conducted. These indicate that Baden (M = 441.41, 95% CI [387.97, 494.85]) and Downstream Craigbourne Dam (M = 564.05, 95% CI [536.18, 591.92]) showed significantly variation from Richmond (M = 859, 95% CI [793.12, 925.92]) and White Kangaroo Rivulet (M = 838.48, 95% CI [736.58, 940.38]), $p<0.001$. Comparison between White kangaroo and Richmond station were not statically significant at $p<0.05$ (Appendix 1.5).

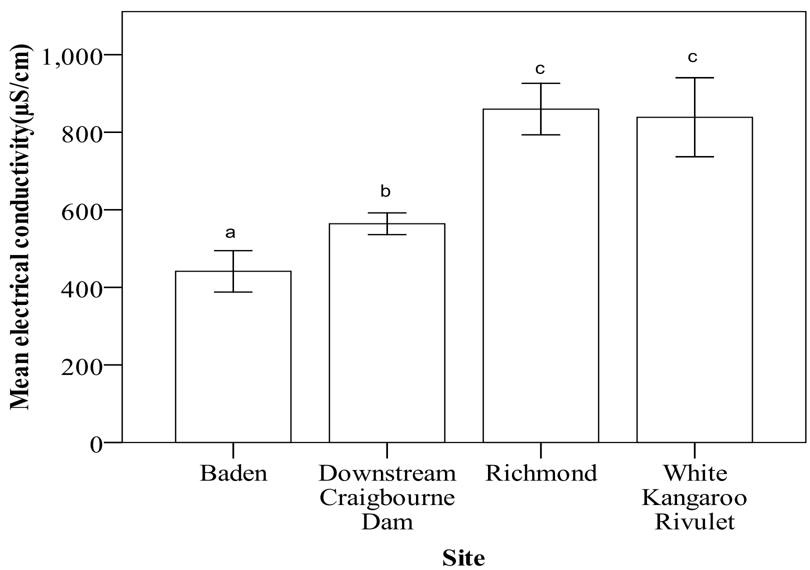


Figure 23 Electrical conductivity data collected at four monitoring stations between 1999 and 2008. Error bars indicate 95% confidence interval of mean. Lower case letters denote results of Games-Howell test ($p<0.05$), used for a post hoc comparisons between means.

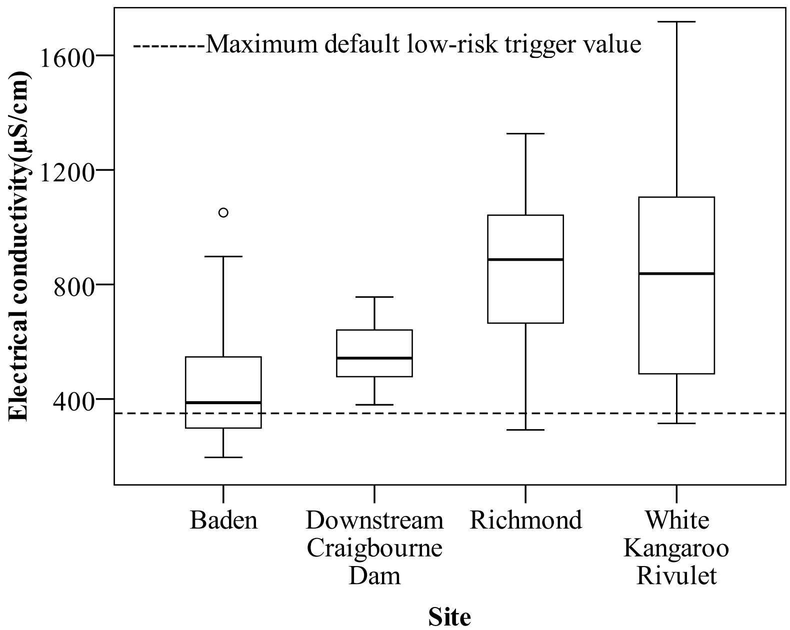


Figure 24 Variability in electrical conductivity data collected at four monitoring stations between 1999 and 2008, with the dotted line representing ANZECC 2000 default low risk trigger values for upland rivers. DPIW 2008 site specific trigger values (80th percentile) are Baden 558 µS/cm, downstream of Craigbourne Dam 509 µS/cm, Richmond 981 µS/cm and White Kangaroo 1146 µS/cm.

Figure 24 shows the distribution of the data for the studied sites. Median electrical conductivity at the Richmond was very high compared to the downstream of Craigbourne Dam and Baden sites while White Kangaroo Rivulet shows the highest electrical conductivity level (1718 µS/cm) recorded on December 6, 2007. Mean monthly electrical conductivity values at Richmond were higher than those from downstream of Craigbourne Dam for all months (Figure 25).

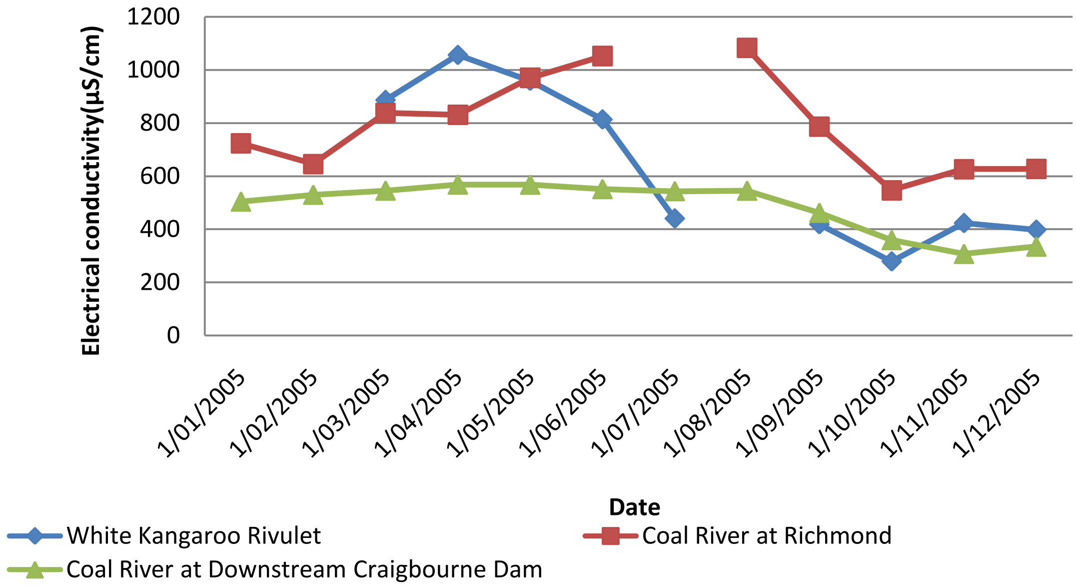


Figure 25 Mean monthly variation of electrical conductivity values at selected station in 2005.

Electrical conductivity of water is a surrogate measure of the amount of dissolved salts. During the summer season or dry periods and when flows are lower, conductivity represents the ground water salinity levels but in high flows periods after rainfall the conductivity more typically represents the dissolved material in the runoff and surface drainage (Bobbi 1998). The data reveals that the conductivity is higher than the National Guidelines recommendation which is 90-350 µS/cm for the upland Tasmanian river water (ANZECC 2000). The electrical conductivity showed the gradual increase from the upstream to downstream stations along the river. These trends could be the attributed to the cumulative impacts of irrigated land use, chemical fertilisers use, low river flows and effluent water discharges from the adjacent town that are characteristic along the Coal River progressing downstream. The river at Richmond is dominated by very intensive agriculture with extensive agrochemical use and irrigation runoff. The chemical fertilisers used on agricultural land leached to the waterways in rainfall and irrigation runoff may be the cause of high values of electrical conductivity. Finnigan (1995) identified the Coal Valley as a high risk area for salinization where more than 15% of farm dams present in the area showed high conductivities (>4 dS/cm) in summer. In addition, soil surface salinity and salt scalds were identified in the Pages Creek subcatchment in the Coal River Valley where highly saline groundwater ranging between 2,102 and 11,922 mg/L was observed (Todd 1999). A study conducted in 2000, with aspects of aquatic ecology of rivers within the Coal River catchment, showed more than 1000 µS/cm EC in Inverquharity Rivulet at Prosser road and Native Hut Rivulet upstream of Campania during spring season (DPIPWE 2003b). These saline rivulets drain into the Coal River above the Richmond weir and could be one of the reasons for the higher electrical conductivity values recorded at the Richmond station. Similarly, White Kangaroo Rivulet showed high electrical conductivity that may be due to the highly saline ground water and effect of the bedrock structures and composition where mafic rock such as dolerite on the western valley side contacts with siliceous Triassic and Permian sedimentary rocks on the eastern valley side (Figure 6). Grose (2003) reported that saline dam water, high soil salinity and saline ground water may be the cause of the moderate stream salinity in the Coal River.

5. 1. 5 Water pH

The data in Figure 26 shows a significant difference in water pH between two of the four gauging stations. While values at Richmond and White Kangaroo Rivulet shows non significance difference between these two stations, those at Baden are significantly lower and downstream of Craigbourne Dam significantly higher than the other sites. Welch F-ratio [F (3, 108.08) = 35.87, $p<0.001$] indicating that mean pH value were significantly different between the stations. Games-Howell post hoc comparisons shows that downstream of Craigbourne Dam (M = 8.25, 95% CI [8.14, 8.35]) had significantly higher pH than Baden (M = 7.40, 95% CI [7.25, 7.55]), Richmond (M = 7.77, 95% CI [7.69, 7.86]) and White Kangaroo Rivulet (M = 7.69, 95% CI [7.64, 7.74]), $p<0.001$. Similarly, Baden mean pH value is significantly lower than Richmond ($p<0.001$) and White Kangaroo Rivulet (<0.05). The comparison also shows that the pH at White Kangaroo Rivulet and Richmond are not significantly different at $p<0.05$ (Appendix 1.6). The box plot (Figure 27) shows that the water in Coal River is highly alkaline downstream of Craigbourne Dam with a median value greater than 8 while the Baden monitoring station at the top of the river has the lowest pH value at 6.6 in September 6, 2005 as compared to the other sites.

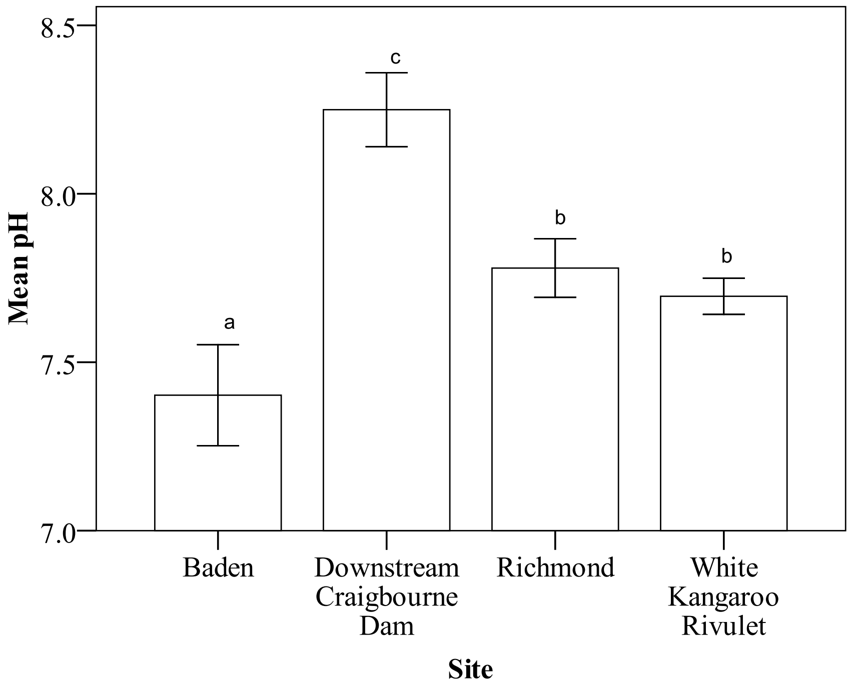


Figure 26 Mean pH values collected at four monitoring stations between 1999 and 2008. Error bars indicate 95% confidence interval of mean. Lower case letters denote results of Games-Howell test ($p<0.05$), used for a post hoc comparisons between means.

The water pH range is variable both seasonally and diurnally depending upon environmental and biological conditions (UNESCO 1992). The box plot shows that pH downstream of Craigbourne Dam is significantly higher than the others sites (Figure 27). High pH value in dams or aquaculture is a result of excess photosynthesis due to aquatic algae (Zweig *et al*. 1999). Similarly, research on sea water conducted in the Florida Bay in the United States of America and Haifa Bay in Israel showed that an increase in the phytoplankaton population in water caused high pH value due to high photosynthetic activity (Fourqurean *et al*. 1993; Kress & Herut 1998). In general, carbon dioxide produced by aquatic organisms during respiration produces an acidic reaction in water. During the day time pH in dams or ponds increases as CO_2 is removed by plants through photosynthesis but at night plants and animals consume oxygen and produce CO_2 which reacts with water and pH levels drop. Apart from the photosynthetic activities, use of chemical fertiliser in farming practices can increase acidity in adjacent river system (Mayo & Noike 1996). While the generally lower pH values at the upstream station (Baden) may have resulted from the dominance of Permian rocks and the presence of improved pastures on these acidic and sodic soils. However, pH values recorded in this station were within the site specific trigger values for sites monitored under the DPIW Baseline Water Quality Monitoring Program (2003-2006) for the Coal River at Baden with 20th- 80th percentile value of 6.7-7.6 (DPIW 2008). The acidity and alkalinity also partly determined by the buffering of rainwater by contact with soil and bedrock (Johnson *et al.* 1972). It means the more time water spends in contact with soil or bedrock the more time there is chemical reactions to occurs. So low stream flow in dry seasons and water collected in dams allows enough time for chemical reactions which can increase pH values (Silsbee & Larson 1982). Similarly pH is also high in increased denitrification conditions in the water (Zilberbbran *et al.* 2001).

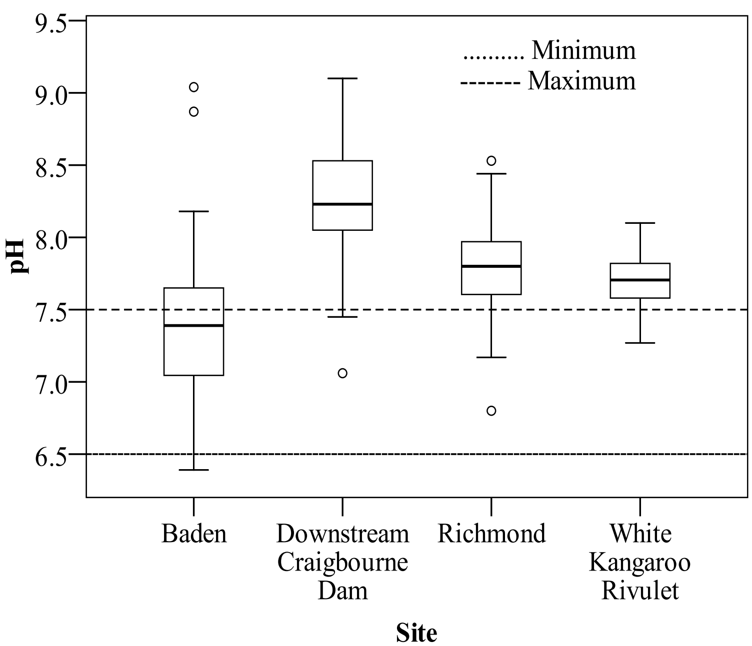


Figure 27 Variability in pH readings at four monitoring sites between 1999 and 2008 and dotted line showing default low- risk trigger values for upland rivers. DPIW 2008 site specific trigger values (80th percentile) are Baden 7.6, downstream of Craigbourne Dam 8.6, Richmond 8.0 and White Kangaroo Rivulet 7.8.

Therefore it is clear several factors such as the role of phytoplankton activity in the dam, riparian land use (sediment and fertiliser input), soil types and their parent rocks, river discharge rates and mineral weathering may be responsible for spatial variation observed in the pH of the river system (Meybeck & Helmer 1989; Kilham 1990).

5. 1. 6 Nutrients

a) Nitrate

A one-way ANOVA was used to examine variation in mean nitrate concentration among the four gauging stations. The confidence intervals shown in Figure 28 indicate that White Kangaroo Rivulet had significantly higher mean values than the other stations. Richmond was not significantly different from Baden and downstream of Craigbourne Dam, but downstream of Craigbourne Dam was significantly higher than Baden. Welch F-ratio showed there was significant variation in mean nitrate between the stations, $F(3, 92.15) = 21.12$, $p<0.001$. To find where the variation lay, Games-Howell post hoc comparisons test were conducted. This revealed that mean nitrate concentrations at White Kangaroo Rivulet (M = 0.18, 95% CI [0.12, 0.24]) were significantly higher than those downstream of Craigbourne Dam (M = 0.06, 95% CI [0.04, 0.08]), Baden (M = 0.006, 95% CI [0.001, 0.11]) and Richmond (M = 0.03, 95% CI [0.13, 0.05]), $p<0.001$.

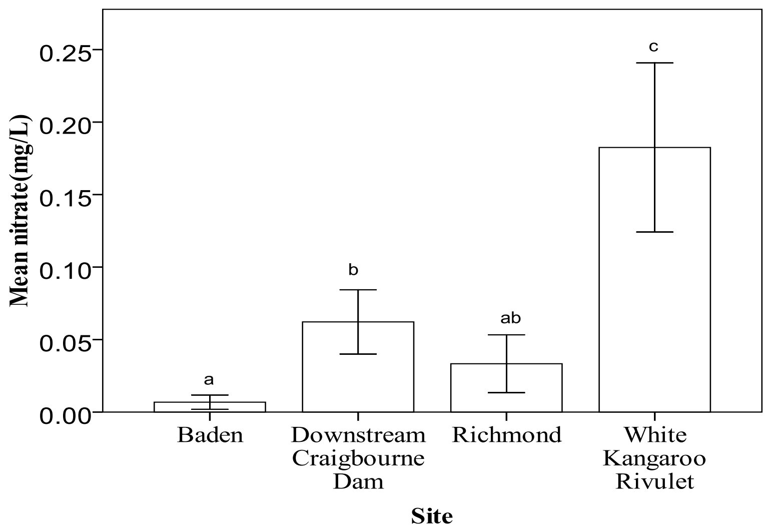


Figure 28 Mean nitrate concentrations collected at four monitoring stations between 1999 and 2008. Error bars indicate 95% confidence interval of mean. Lower case letters denote results of Games-Howell test ($p<0.05$), used for a post hoc comparisons between means.

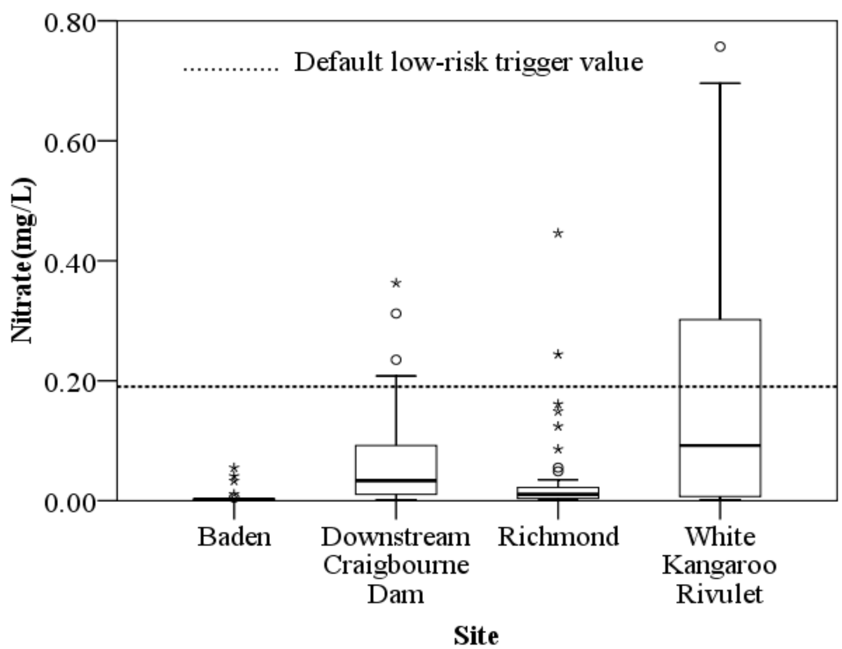


Figure 29 Variability in nitrate concentrations at four monitoring sites between 1999 and 2008 with and dotted line represent ANZECC 2000 default low risk trigger values for upland rivers in Tasmania. DPIW 2008 site specific trigger values (80th percentile) are Baden 0.003 mg/L, downstream of Craigbourne Dam 0.074 mg/L, Richmond 0.021 mg/L and White Kangaroo Rivulet 0.476 mg/L.

b) Total nitrogen

The plot of mean total nitrogen values in Figure 30 indicates that the values from downstream of Craigbourne Dam were significantly higher than Richmond station. Welch F-ratio showed significant variation between stations, F (3, 85.67) = 6.20, p<0.001. However, the highest mean values were in White Kangaroo Rivulet (M = 0.90 95% CI [0.66, 1.13]) but not significant different with other stations. Games-Howell post hoc multiple comparison reveal Richmond (M = 0.70, 95% CI [0.64, 0.76]), had significantly lower values than downstream of Craigbourne Dam (M= 0.86, 95% CI [0.81, 0.91]) the, p<0.001 and there was no significant difference observed in others stations at p<0.05 (Appendix 1.7).

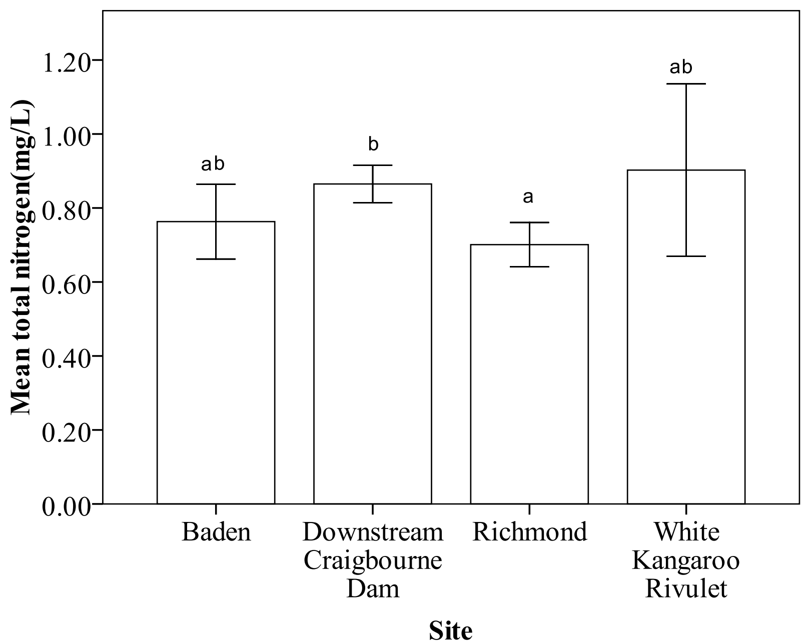


Figure 30 Mean total nitrogen data collected at four monitoring stations between 1999 and 2008. Error bars indicate 95% confidence interval of mean. Lower case letters denote results of Games-Howell test (p<0.05), used for a post hoc comparisons between means.

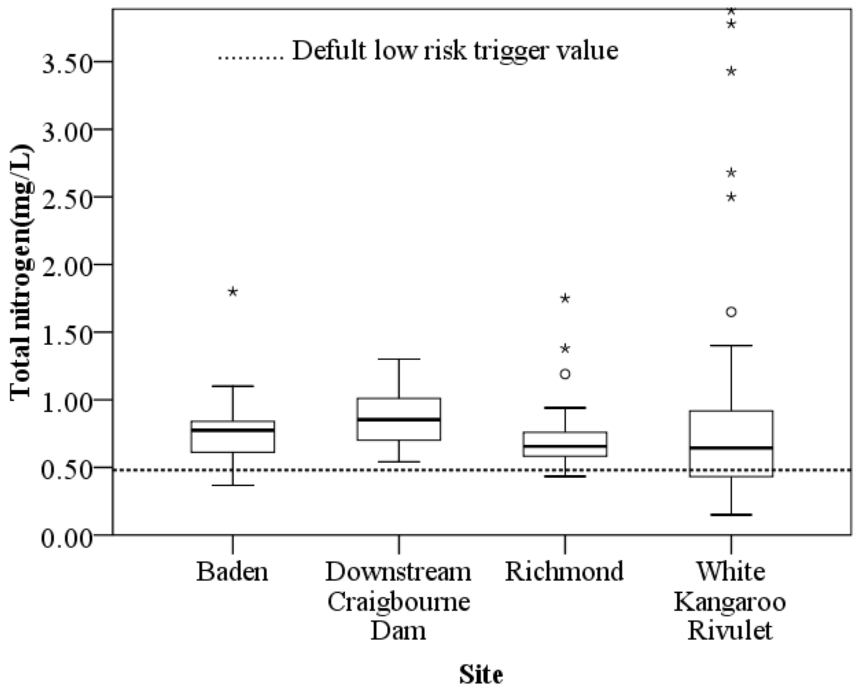


Figure 31 Variability in total nitrogen concentrations at four monitoring sites between 1999 and 2008 with the dotted line represents ANZECC 2000 low–risk trigger values for Upland River. DPIW 2008 site specific trigger values (80th percentile) are Baden 0.838 mg/L, downstream of Craigbourne Dam 0.923 mg/L, Richmond 0.783 mg/L and White Kangaroo Rivulet 0.913 mg/L.

c) Dissolved reactive phosphorus(DRP)

Figure 32 shows values for mean of dissolved reactive phosphorus collected at each station with 95% confidence intervals. This shows that values at Baden were significantly lower than those from downstream of Craigbourne Dam and White Kangaroo Rivulet but not with Richmond. Welch F-ratio showed there was significant variation in mean dissolved reactive phosphorus between the stations, F (3, 101.71) = 14.84, p<0.001. Games-Howell post hoc test revealed that mean dissolved reactive phosphorus at Baden (M = 0.003, 95% CI [0.002, 0.003]) was significantly lower than downstream of Craigbourne Dam (M = 0.007, 95% CI [0.008, 0.006]), and White Kangaroo Rivulet (M = 0.006, 95% CI [0.004, 0.007]), but not Richmond (M = 0.004, 95% CI [0.003, 0.005]), p<0.001. Similarly, Richmond was significantly lower with Downstream Craigbourne

Dam (p = 0.005) (Appendix 1.7). Box plot shows the distribution of collected sample among the gauging stations (Figure 33).

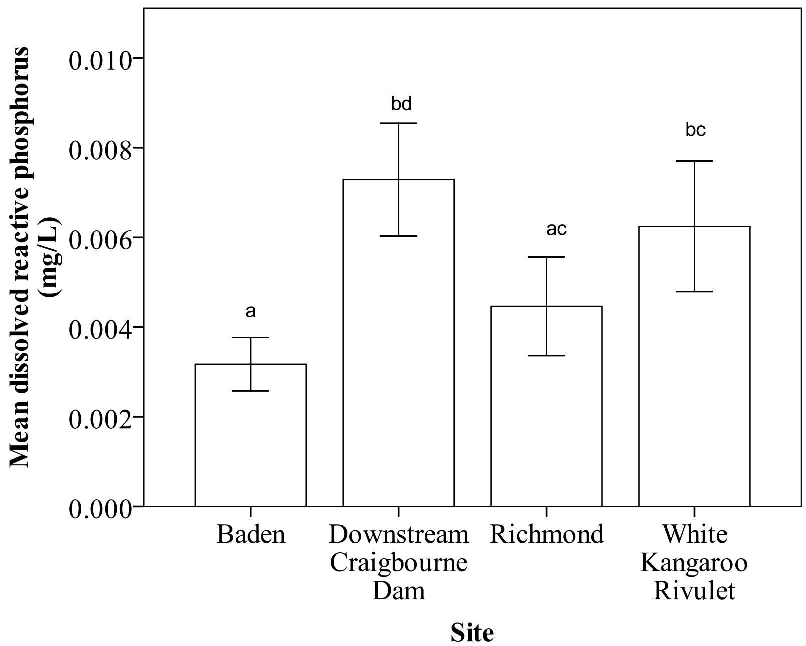


Figure 32 Error bar chart of the dissolved reactive phosphorus data collected at various monitoring stations during period of 1999-2008. Error bars indicate 95% confidence interval of mean. Lower case letters denote results of Games-Howell test ($p<0.05$), used for a post hoc comparisons between means.

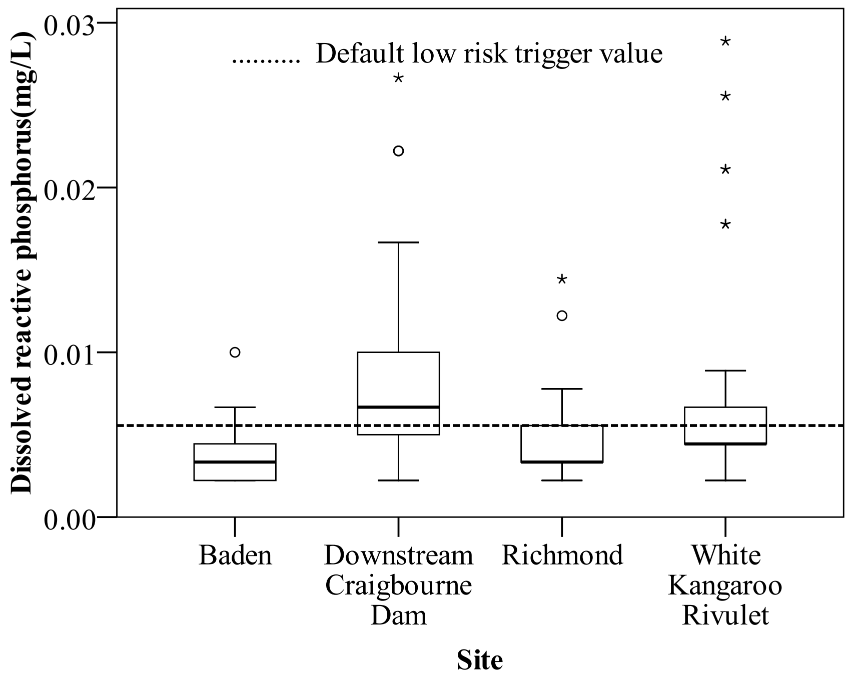


Figure 33 Variability in dissolved reactive phosphorus concentrations at four monitoring sites during 1999-2008 and dot line represents ANZECC 2000 default low-risk trigger values for Upland River in Tasmania. DPIW 2008 site specific trigger values (80th percentile) are Baden 0.004 mg/L, downstream of Craigbourne Dam 0.007 mg/L, Richmond 0.004 mg/L and White Kangaroo Rivulet 0.006 mg/L.

d) Total phosphorus

Figure 34 shows mean total phosphorus data. The mean values seem to indicate that Downstream of Craigbourne Dam had higher total phosphorus concentration than the other sites. However, the Welch F-ratio showed mean total phosphorus was only significantly varied between some stations, $F(3, 87.32) = 3.49$, $p = 0.019$. Multiple comparisons of mean (Games-Howell post hoc test) showed that downstream of Craigbourne Dam (M = 0.031, 95% CI [0.27, 0.35]) was significantly higher than Richmond (M = 0.019, 95% CI [0.01, 0.02]), $p = 0.009$ but not significantly different to the other stations at $p<0.05$ (Appendix 1.8).

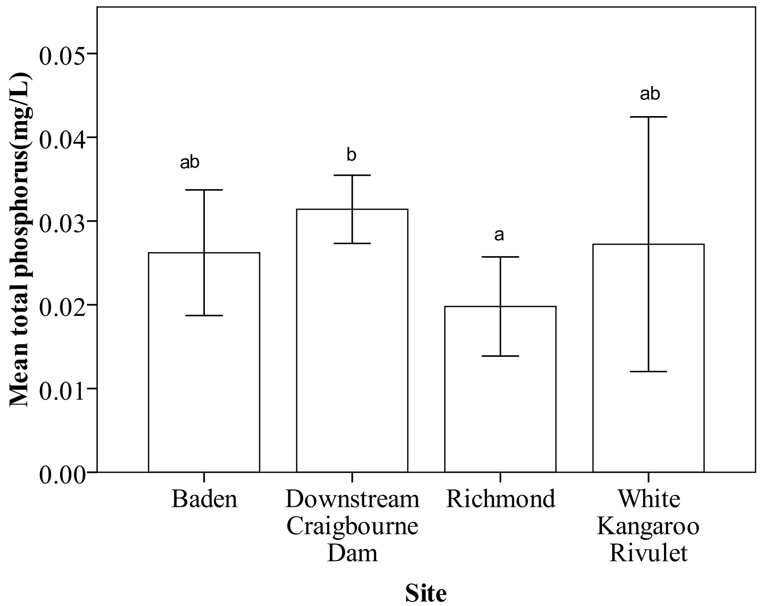


Figure 34 Mean total phosphorus data collected at four monitoring stations between 1999 and 2008. Error bars indicate 95% confidence interval of mean. Lower case letters denote results of Games-Howell test ($p<0.05$), used for a post hoc comparisons between means.

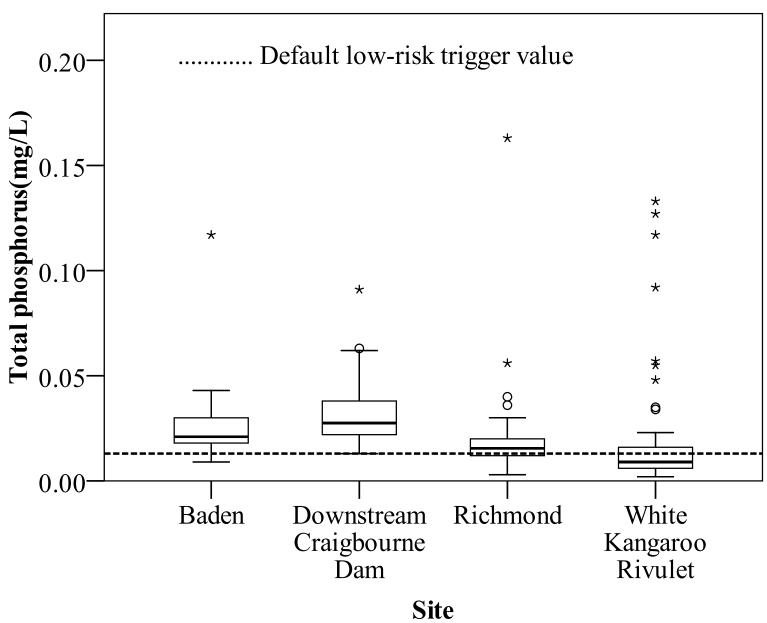


Figure 35 Variability in total phosphorus values collected at four monitoring sites between 1999 and 2008 with the dotted line represents ANZECC 2000 default low-risk trigger values for upland rivers in Tasmania. DPIW 2008 site specific trigger values (80th percentile) are Baden 0.033 mg/L, downstream of Craigbourne Dam 0.030 mg/L, Richmond 0.024 mg/L and White Kangaroo Rivulet 0.015 mg/L.

Nutrient concentrations in the river water are indicative of water quality. Analysis of variance indicated some significant variations in nutrients concentration between the sampling sites. The standard nitrate nitrogen concentration for natural river systems is less than 0.1 mg/L (UNESCO 1992). If the concentration is above this level, it suggests the influence of several activities such as the presence of fertiliser, industrial and municipal waste water, and land clearing (Stevens & Hornung 1988) and agricultural activities in general including grazing (Ferrier *et al.* 2001). Among the four sites, only the mean nitrate values at White Kangaroo Rivulet monitoring station were significantly higher than the other stations (Figure 24). In terms of phosphorous concentrations, Reddy *et al.* (1999) reported that dissolved and particulate phosphorus can also be accumulated in sediments of impoundment or small reservoirs, while Hannan *et al.* (1972) and Neary *et al.* (2010) reported that man-made dam or reservoirs in a river can act as the nutrient traps. This may explain why higher phosphorus concentrations were recorded at downstream of Craigbourne Dam than at the other sites. Another common source for higher median nutrients concentration at Craigbourne Dam could be the presence of waste water lagoons at Colebrook which leach treated effluent into waterway. Bobbi (1997b) reported that higher concentration of nitrogen and phosphorus in Wallby Rivulet could be one of the factors for increased nutrient concentration in the Craigbourne Dam. He thought nutrients were leached to the Wallby Rivulet through surface flow from the Colebrook sewage treatment plant. Similarly, Baker (2000) reported that point sources of erosion were the main contributors of sediment and that sediments were the source of the phosphorus and nitrogen in Craigbourne Dam. He found total estimated mass of phosphorus (45 tonnes) and nitrogen (245 tonnes) collected in dam floor sediment since dam construction in 1986. This represents an annual inflow of 3.2 tonnes of P and 17.5 tonnes of N per year. This could well be one of the reasons for the higher median concentrations of phosphorus and nitrogen in the Craigbourne Dam. However, mean phosphorus concentrations from all stations were well above the ANZECC 2000 low risk trigger value of 0.013 mg/L for agricultural catchments in Tasmania (DPIW 2008). The concentration of total phosphorus lower than 0.01 mg/L is considered low for the Tasmanian agricultural catchment (Bobbi 1998).

5. 1. 7 Flow

Mean flow at four sites were recorded in the range of 0.05 to 0.2 cubic metres per second (Figure 36). Welch F-ratio showed there was a significant spatial effect in the observed data, $F(3, 91.39) = 4.29$, $p = 0.007$. For multiple comparisons of means a Games-Howell post hoc test was conducted. This showed that downstream of Craigbourne Dam ($M = 0.194$, 95% CI [0.12, 0.26]), had significantly higher values than Baden ($M = 0.05$, 95% CI [0.0004, 0.11]), and White Kangaroo Rivulet ($M = 0.06$, 95% CI [0.002, 0.12]), $p = 0.007$ but was not significantly difference from flow at Richmond ($M = 0.16$, 95% CI [0.007, 0.32]) $p<0.05$.

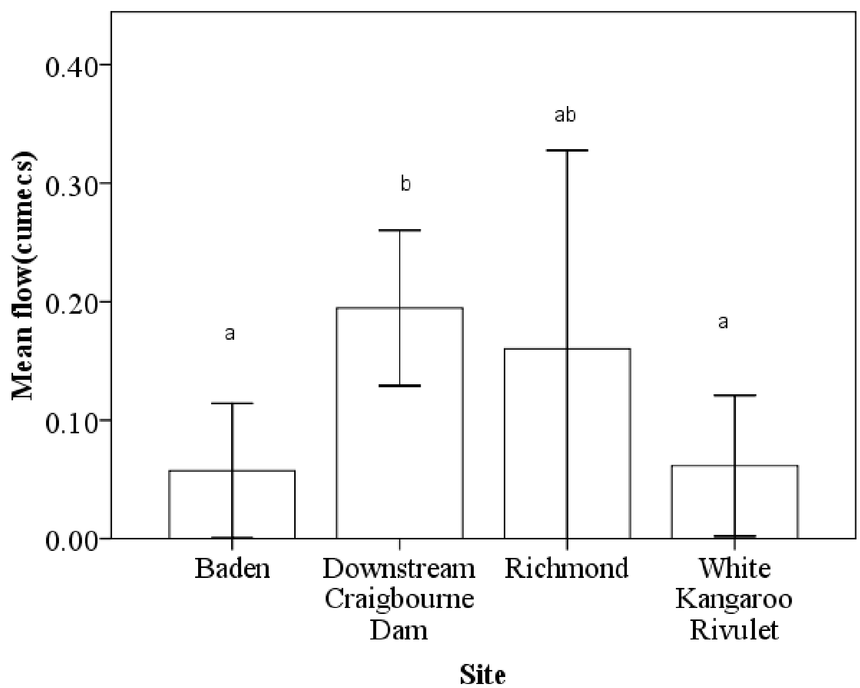


Figure 36 Mean flow data collected at four monitoring stations between 1999 and 2008. Error bars indicate 95% confidence interval of mean. Lower case letters denote results of Games-Howell test (p<0.05), used for a post hoc comparisons between means.

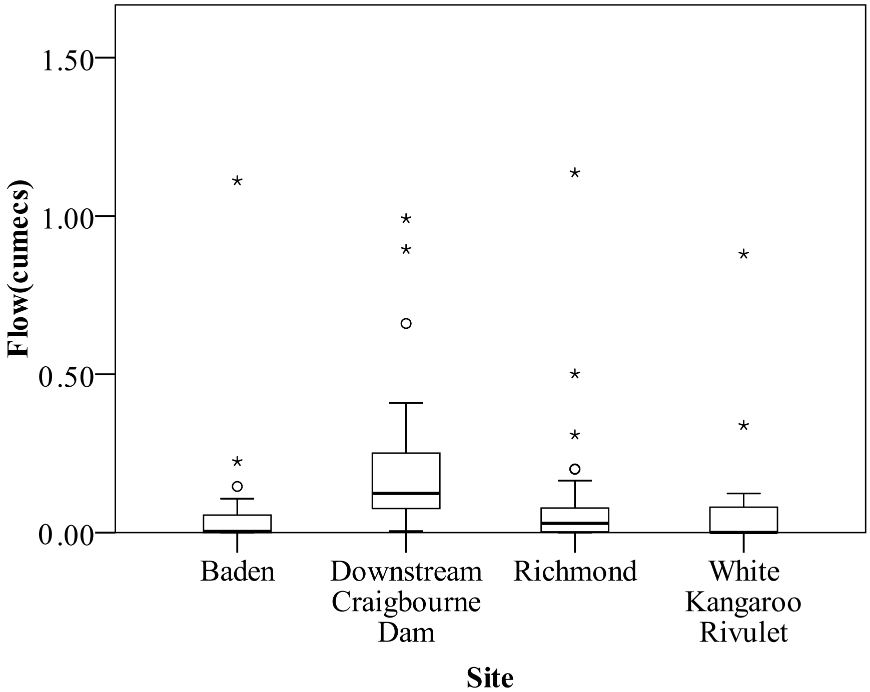


Figure 37 Variability in stream flow at four monitoring sites between 1999 and 2008.

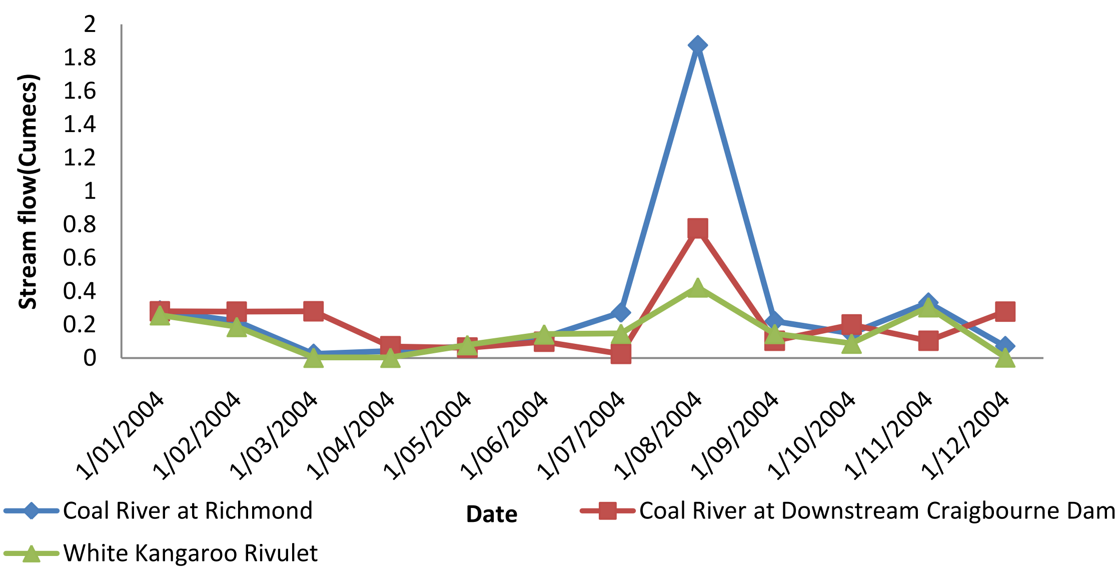


Figure 38 Monthly variation of stream flow in the river system in 2004.

The median data however showed that flow downstream of Craigbourne Dam was higher than that at the other stations (Figure 37).

The data showed that stream flow was higher at Richmond only in the winter and in summer but rest of the periods downstream of Craigbourne Dam showed the highest flow in 2004. On the other hand White Kangaroo Rivulet had lower stream flow compared with the other stations in most seasons. Direct extraction of water for irrigation during the summer seasons is most likely to be the cause of this lower flow. The higher flow downstream of Craigbourne Dam appears to be the result of water accumulated in the Dam being released continuously downstream (Figure 38).

5. 1. 8 Total nitrogen and total phosphorus ratio

The ratios of total nitrogen to total phosphorus indicate significantly lower mean values at downstream of Craigbourne Dam than Richmond and White Kangaroo Rivulet (Figure 39).

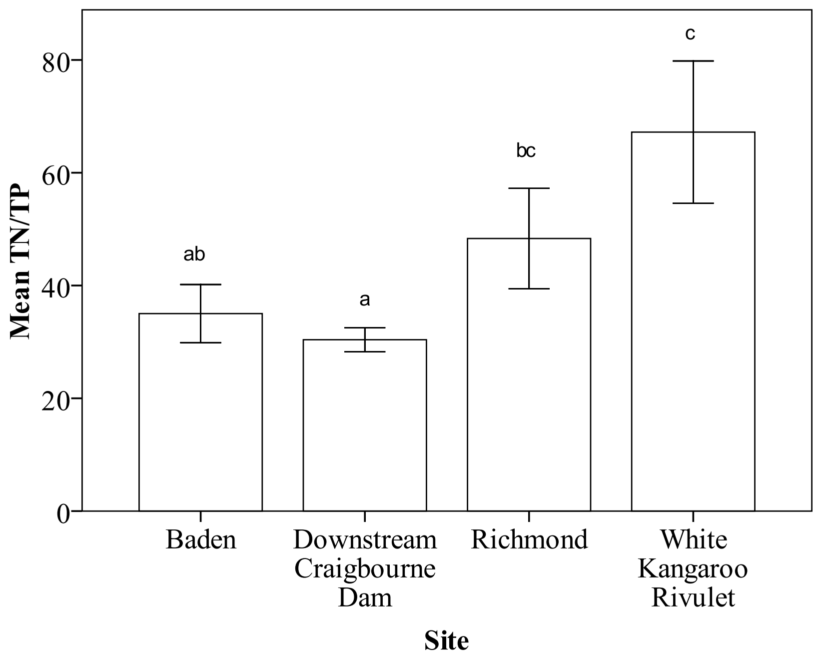


Figure 39 Mean TN/TP ratios for four monitoring stations between 1999 and 2008. Error bars indicate 95% confidence interval of mean. Lower case letters denote results of Games-Howell test ($p<0.05$), used for a post hoc comparisons between means.

Welch F-ratio showed significant variation between some stations, $F(3, 79.65) = 15.78$, $p<0.001$. Games-Howell post hoc comparison of means showed that downstream of

Craigbourne Dam (M = 30.38, 95% CI [28.25, 32.52]) had significantly lower values than Richmond (M = 48.33, 95% CI [39.43, 57.23]), p = 0.018 and White Kangaroo Rivulet (M = 45.74, 95% CI [54.59, 79.81]), p<0.001 but not Baden. The other sites were not significantly different from each other (Appendix 1.9).

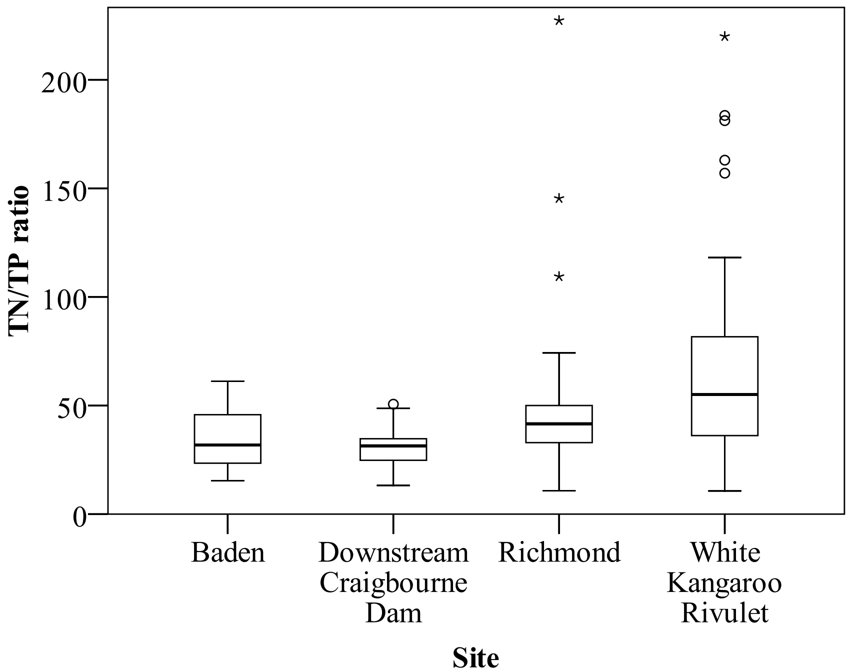


Figure 40 Variability in TN/TP ratios at four monitoring sites between 1999 and 2008.

The TN/TP ratios can be used to determine the nutrient limitations for phytoplankton and periphyton growth in fresh water. If the ratio of TN/TP falls below 5:1 or exceeds 20:1 (by weight), this indicates nitrogen or phosphorus limitations for the phytoplankton growth respectively (Thomann & Mueller 1987). The data here for mean TN/TP ratios (Figure 39) would suggest that Craigbourne Dam has the closest ratio to these optimal values for algal blooms.

5. 2 Interaction between water quality parameters

5. 2. 1 Dissolved oxygen and water pH

The regression analysis revealed a significant relationship ($p<0.05$) between dissolved oxygen and pH in the Coal River at Richmond (Figure 41) but those relationships did not exist at White Kangaroo Rivulet, downstream of Craigbourne Dam or at Baden.

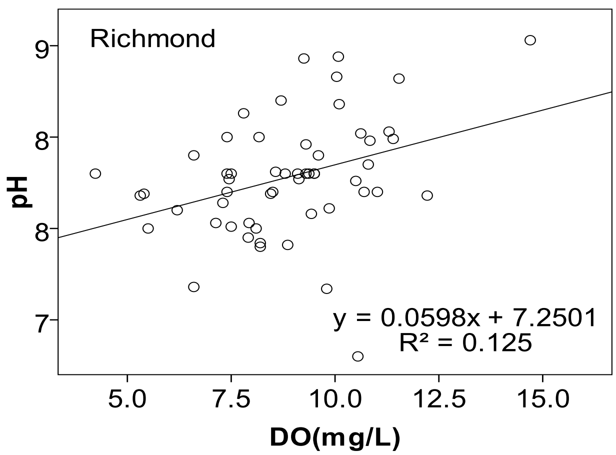


Figure 41 Relationship between dissolved oxygen and water pH at Richmond in the Coal River.

As algae photosynthesize during the day, carbon dioxide is taken up, resulting in a reduction in free hydrogen ions and an increase in water pH. At night, respiration produces CO_2, reducing water pH. Daily fluctuations of dissolved oxygen in impounded waters like ponds and commercial dams are higher than those in the rivers or open sea. These fluctuations are in response to photosynthetic activities of the aquatic plants and respiration of aquatic organisms. So, the lowest levels of dissolved oxygen occur in the morning and rise slowly when plant starts photosynthesis in the afternoon (Boyd 1990). Thus, synchronized fluctuation of dissolved oxygen and pH are result of the photosynthetic and respiration activity of aquatic organism (Pogue & Anderson 1995).

5. 2. 2 Dissolved oxygen and water temperature

Water temperatures in the river system are generally influenced by geology, topography of the region, water flow regime, land use practices, and riparian vegetation. In general the higher the water temperature, the lower the solubility of the oxygen in the water (Braden 2001; Weiss 1970) making temperature one of the driving mechanisms for dissolved oxygen (DO) levels in the river. Because of its relationship with dissolved oxygen, fluctuations in water temperatures are an important concern. Murdoch *et al* (2000) reviewed the warming surface water effect on reduction of dissolved oxygen concentration in water during drought. They found the higher the surface water temperature the lower the dissolved oxygen concentration in the water.

Low stream flow, evaporation and the accumulation of organic matter are also responsible for depletion of oxygen concentration in the water (Justic *et al.* 1997; Bellos & Sawidis 2005). In addition, nutrient concentrations, salinity and photosynthetic rate have significant impacts on dissolved oxygen concentration in water. This inverse relationship between dissolved oxygen and water temperature was observed at all gauging stations except Richmond where the relationship was not significant at $p<0.05$ (Figure 42). Similarly, van Vliet & Zwolsman (2008) have shown that a strong negative relationship between DO and water temperature ($R^2 = 0.76$, $p<0.01$) in the Meuse River.

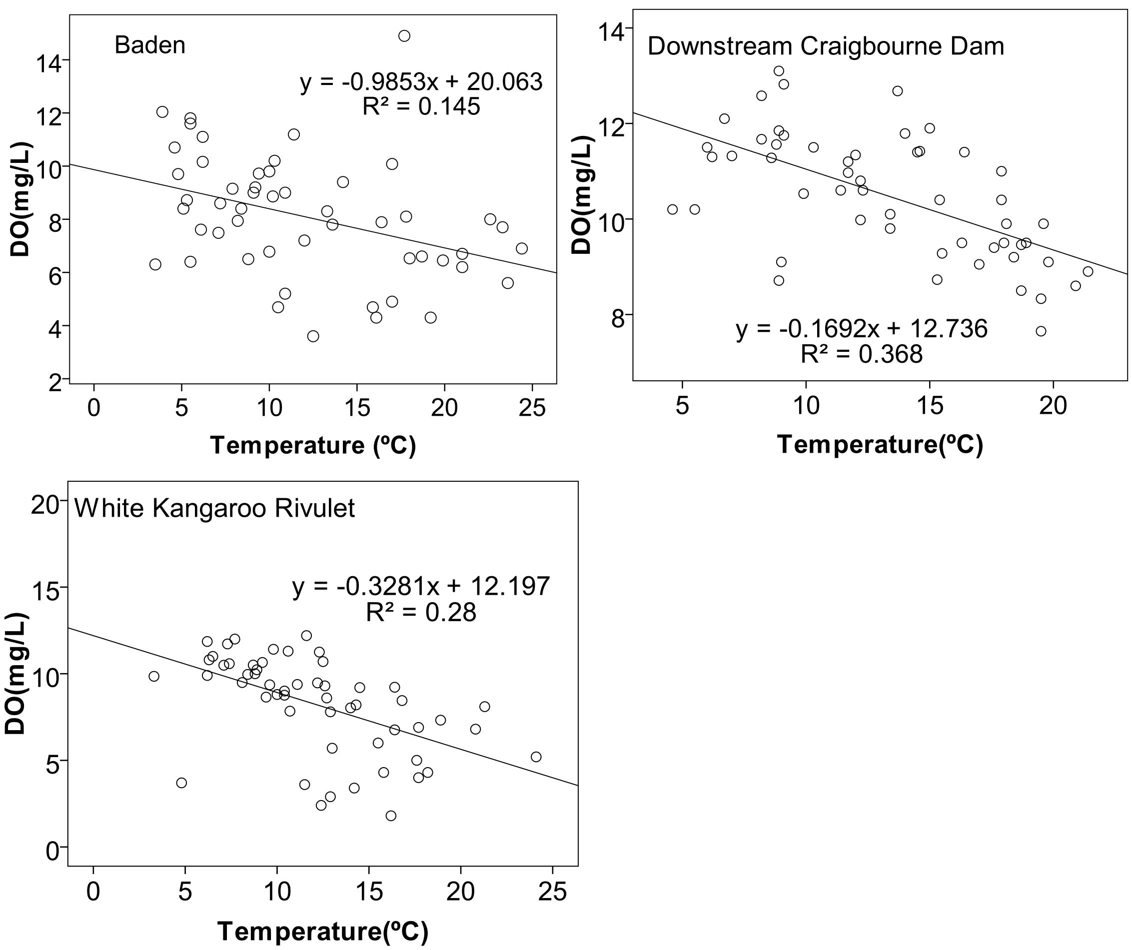


Figure 42 Relationship between dissolved oxygen (DO) and water temperature at downstream of Craigbourne Dam and Baden in the Coal River and White Kangaroo Rivulet.

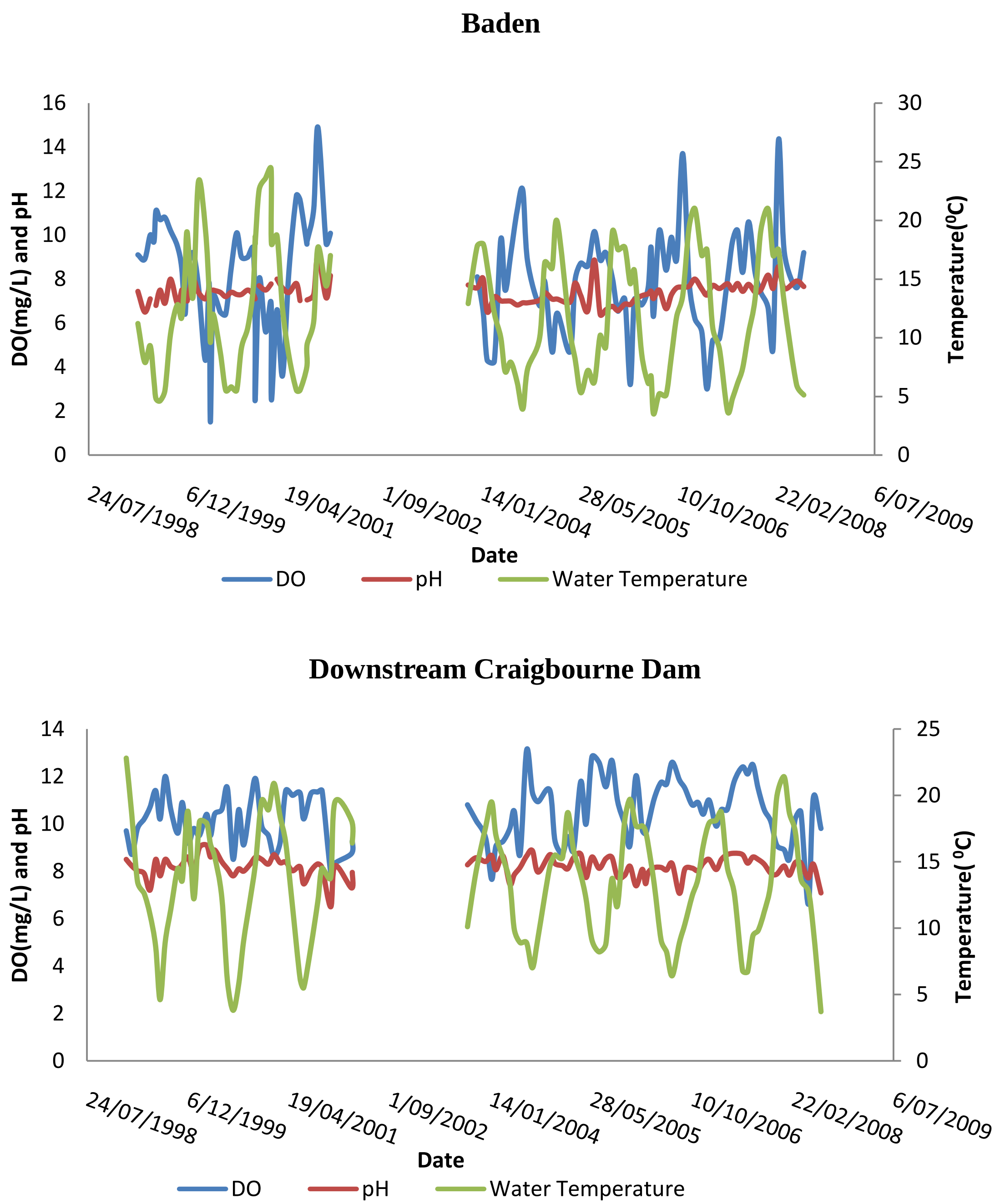
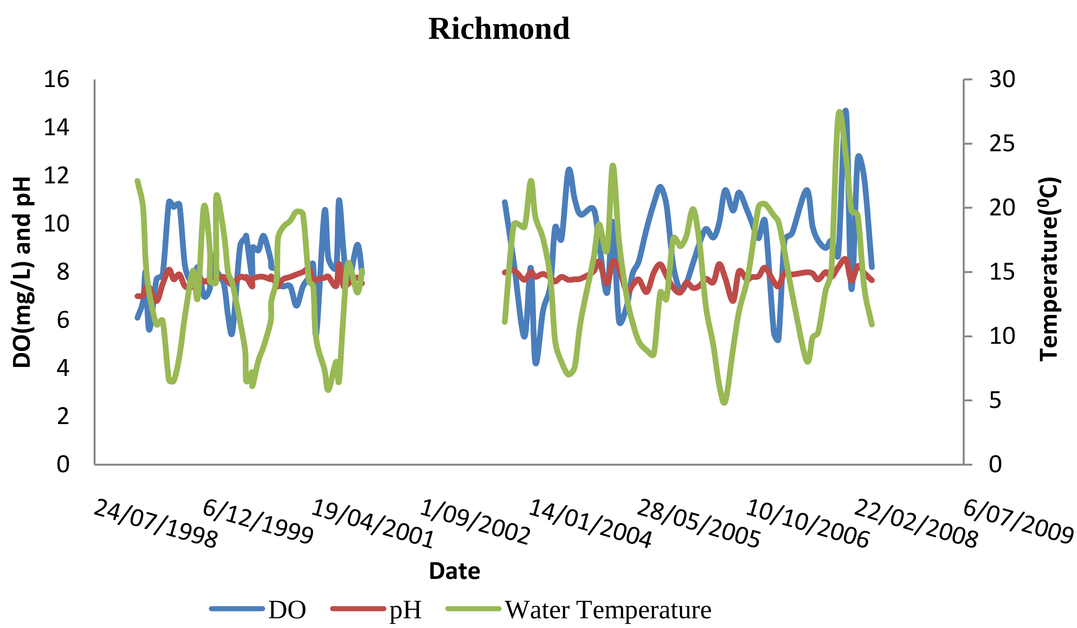


Figure 43 Dissolved oxygen (DO), water pH and temperature relationship in Baden and downstream of Craigbourne Dam water monitoring stations, graphs show the strong inverse relationship between DO and water temperature.



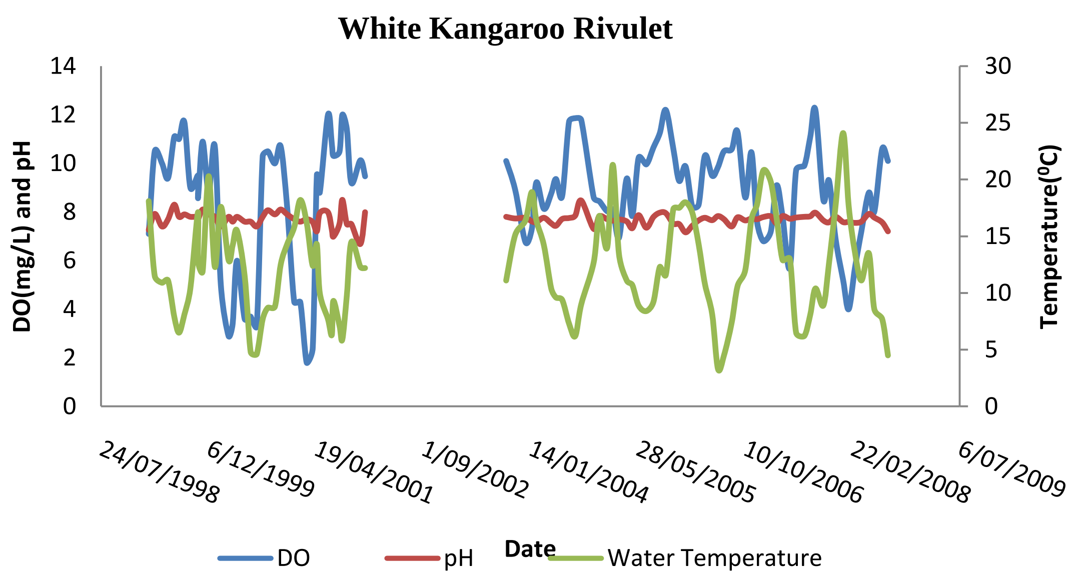


Figure 44 Dissolved oxygen (DO), water pH and temperature relationship in Richmond and White Kangaroo Rivulet water monitoring stations, graphs show the strong inverse relationship between DO and water temperature.

5. 2. 3 Relationship between dissolved oxygen (DO), electrical conductivity (EC) and water temperature

Oxygen dissolves more readily in water with low levels of suspended solids. When salt concentration increase in water bodies through runoff the amount of dissolved oxygen decreases. A significant relationship between dissolved oxygen and electrical conductivity was observed only at Baden and White Kangaroo Rivulet but not the other two stations (Figure 45).

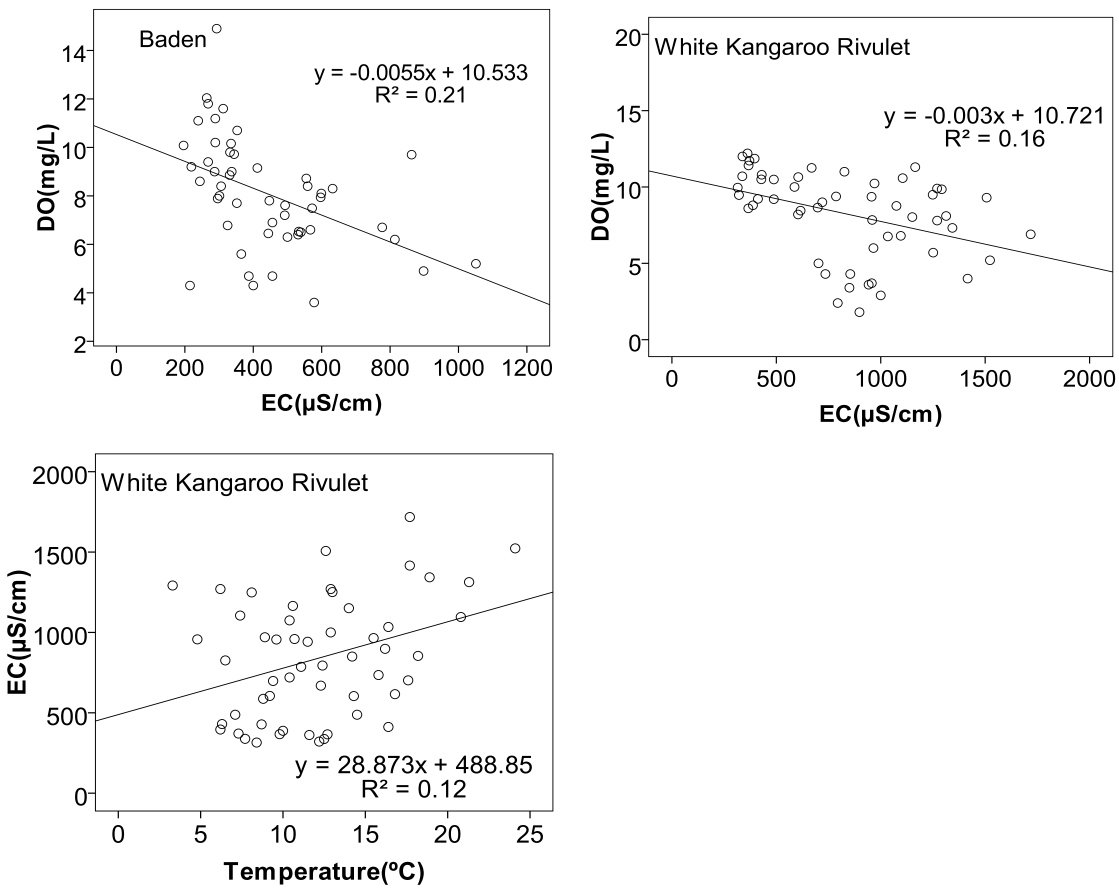


Figure 45 Relationship between dissolved oxygen, electrical conductivity and temperature in the Coal River at Baden and White Kangaroo Rivulet.

The concentration of dissolved oxygen depends on water temperature and salinity (Weiss 1970) where high temperature and salinity reduced the oxygen level in water. Electrical conductivity in water invariably increases with an increase in temperature. Warm water is less viscous and has greater electronic movement, thus allowing free flow of electric current (Weiss 1970). Small variations in temperature can result in marked differences in conductivity. Electrical conductivity is significantly correlated with the water temperature and pH (Boyd & Lichktoppler 1979). So, high water temperature increases EC and decreases the DO in the water.

5. 2. 4 Rainfall effect on turbidity and nutrients

Turbidity levels are generally very low in the Coal River. Rainfall is one of the main factors contributing to suspended solids entering river systems, and significant relationships were found between rainfall and turbidity at Baden, Richmond and White Kangaroo Rivulet but not downstream of Craigbourne Dam (Figure 46). This is most likely due to the role of dams in settling suspended solids before they reach the gauging station. The regression analyses also showed that rainfall was significantly related ($p<0.05$) to nitrate-nitrogen, total nitrogen, dissolved reactive phosphorus (DRP) and total phosphorus in the Coal River water at Richmond (Figure 46) but not at Baden, downstream of Craigbourne Dam or White Kangaroo Rivulet. Significant effects of rainfall on nitrate and total nitrogen suggest flushing of nitrate out of soils following rainfall. The phosphorous in the water is found in two different forms; namely bound and dissolved. The dissolved reactive phosphorus (DRP) is a free form of phosphorus and easily available to aquatic plants. The higher dissolved reactive phosphorus concentration indicates that phosphorus is coming into water from sources such as fertiliser runoff or sewage effluent discharge. The significant relationship between rainfall and dissolved reactive phosphorus in this study suggests that phosphorus has accumulated in the river from the cropping areas during rainfall events. Verhoff *et al*. (1982) reported that higher concentration of phosphorus in river water was recorded during storms. Blanchard & Lerch (2000) reported that nitrate-nitrogen has potential to be washed out or leached and transported in runoff.

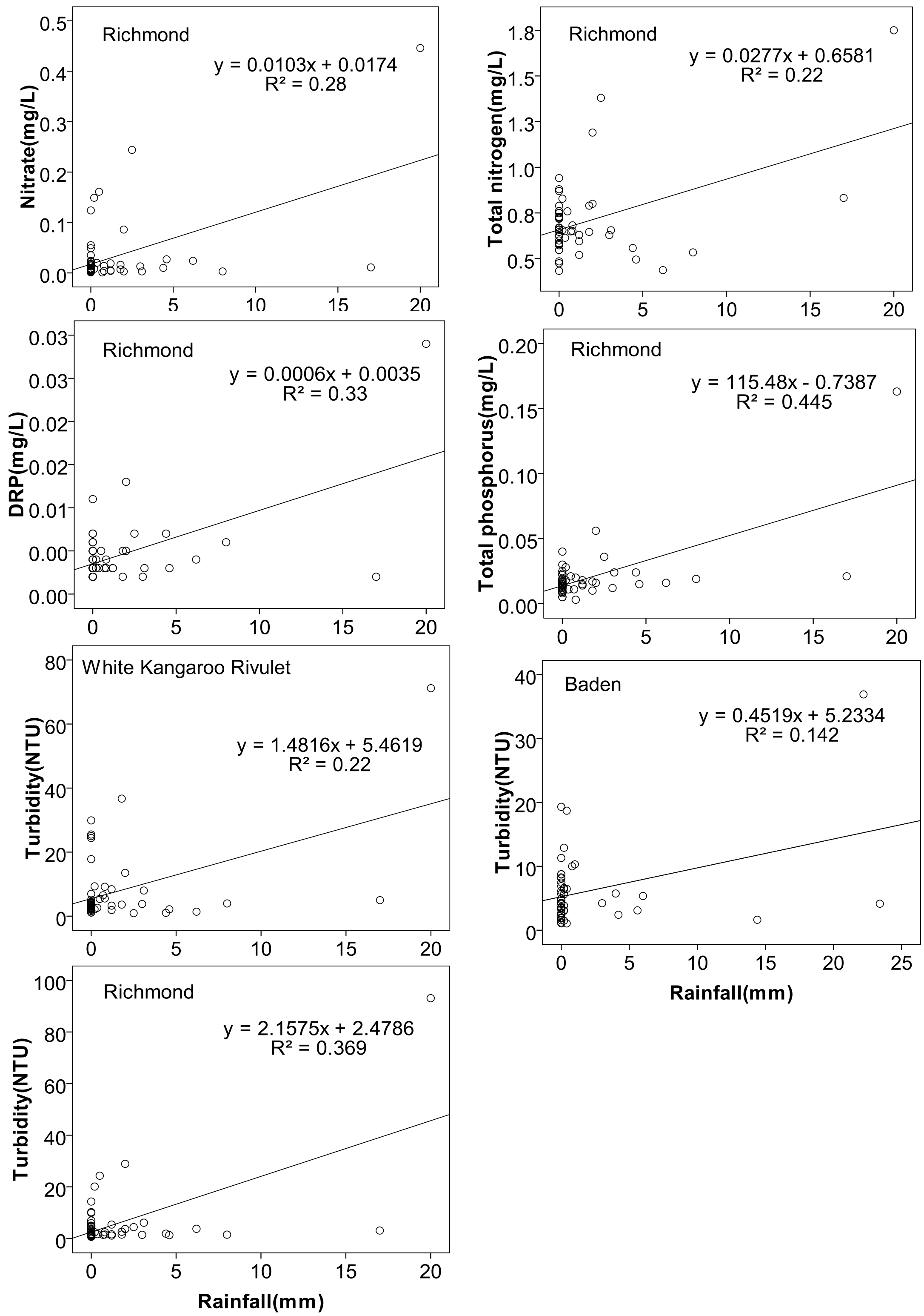


Figure 46 Effect of rainfall on turbidity and nutrients concentration at in the Coal River and White Kangaroo Rivulet.

5. 2. 5 Turbidity and nutrients

Stream turbidity is a strong determinant of total phosphorus in river water because phosphorus is quickly attached to the solid particles found in turbid water (Bobbi 1998). A significant relationship between turbidity and total phosphorus was observed downstream of Craigbourne Dam, Richmond and White Kangaroo Rivulet (Figure 47) but not at Baden. Buckney (1979) reported that suspended solids are an important vehicle for transport of phosphorus. The data also suggest that higher turbidity means higher total phosphorus concentration in the river water. Similar findings have been reported by Bobbi (1998) in Huon River at Judbury in Tasmania with significant relationship between turbidity and total phosphorus concentration (R^2 = 0.89, p<0.001).

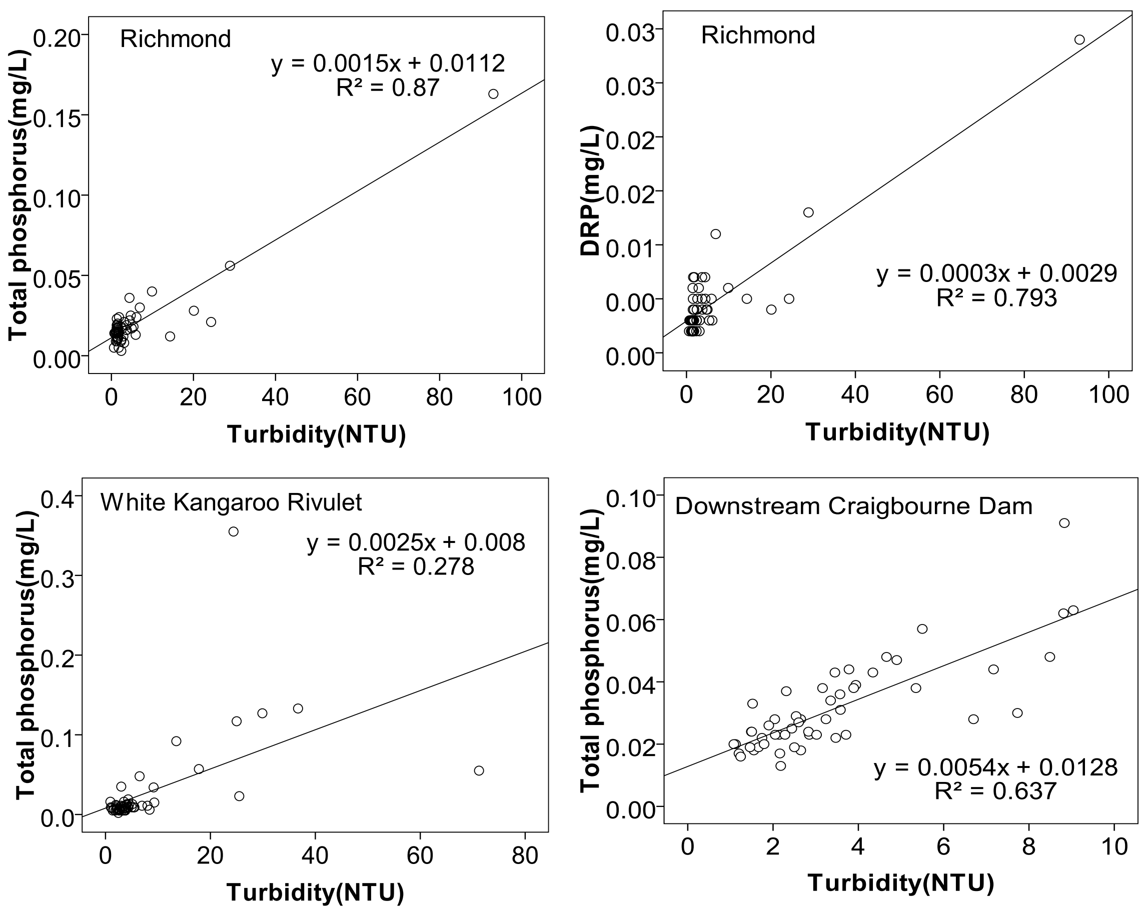


Figure 47 Significant relationships between turbidity and phosphorus concentrations in the Coal River and White Kangaroo Rivulet.

The nitrogen and turbidity relationship for the Coal River and the White Kangaroo Rivulet is defined by the linear regression (which is significant at 0.01 level). There is significant relationship in all gauging station except Baden (Figure 48). This may be due to the poor data set at Baden. Similar finding were reported from Buttons Creek in north-west Tasmania, where total nitrogen and turbidity was significantly correlated (Cotching & Sims 2003).

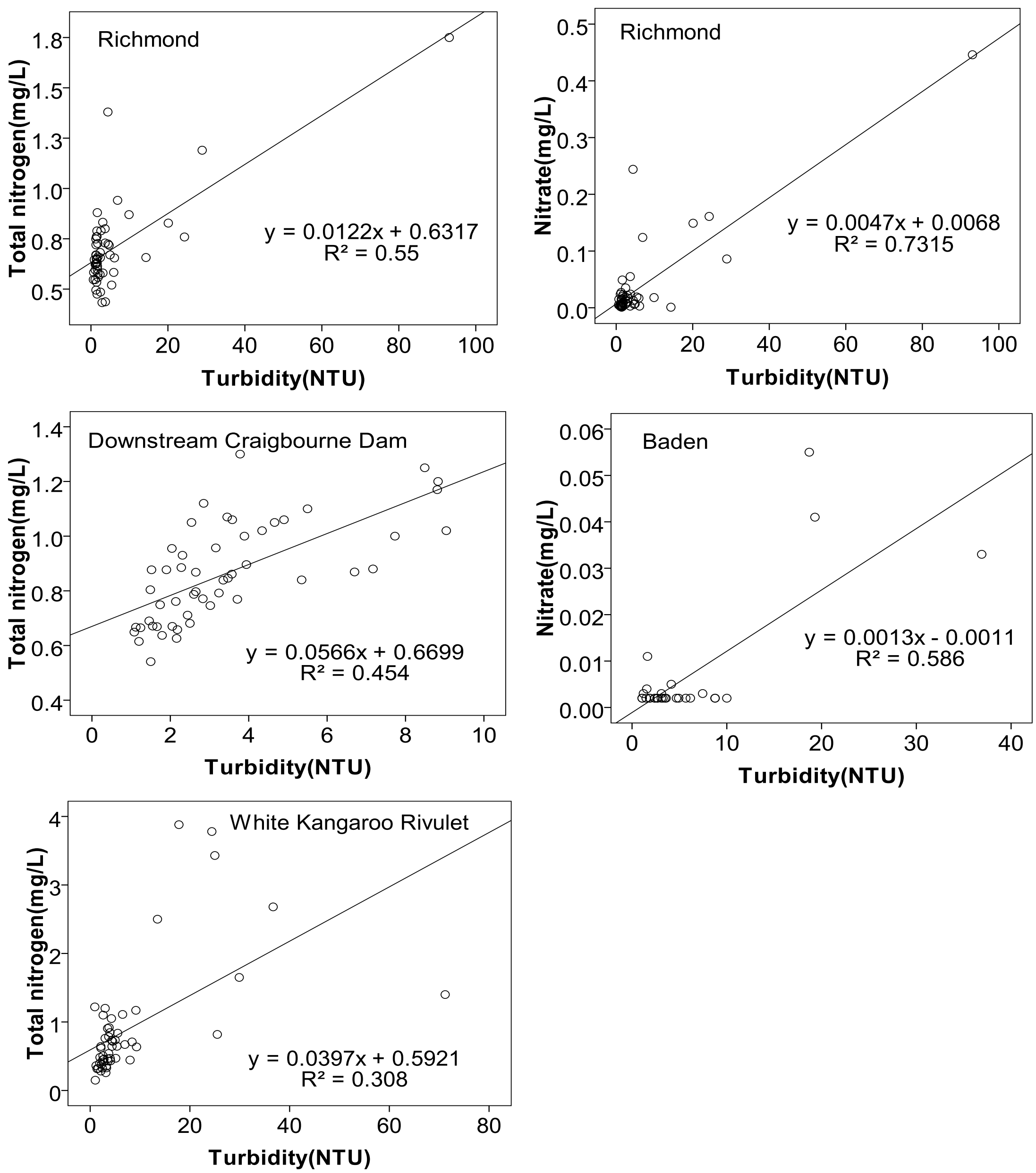


Figure 48 Significant relationships between turbidity and nitrogen concentrations in the Coal River and White Kangaroo Rivulet.

5. 2. 6 Stream flow effect on turbidity and nutrients

Stream flow and turbidity were significantly related at three monitoring stations but not downstream of Craigbourne Dam (Figure 49). The impact of dams or other water impoundments is to allow suspended materials or solid particles carried by the upstream current to settle out of the water column. While turbid water at other stations showed the impact of human activities such as clearing of riparian vegetation, access to water course or river by cattle or sheep, road drainage and waste irrigation water.

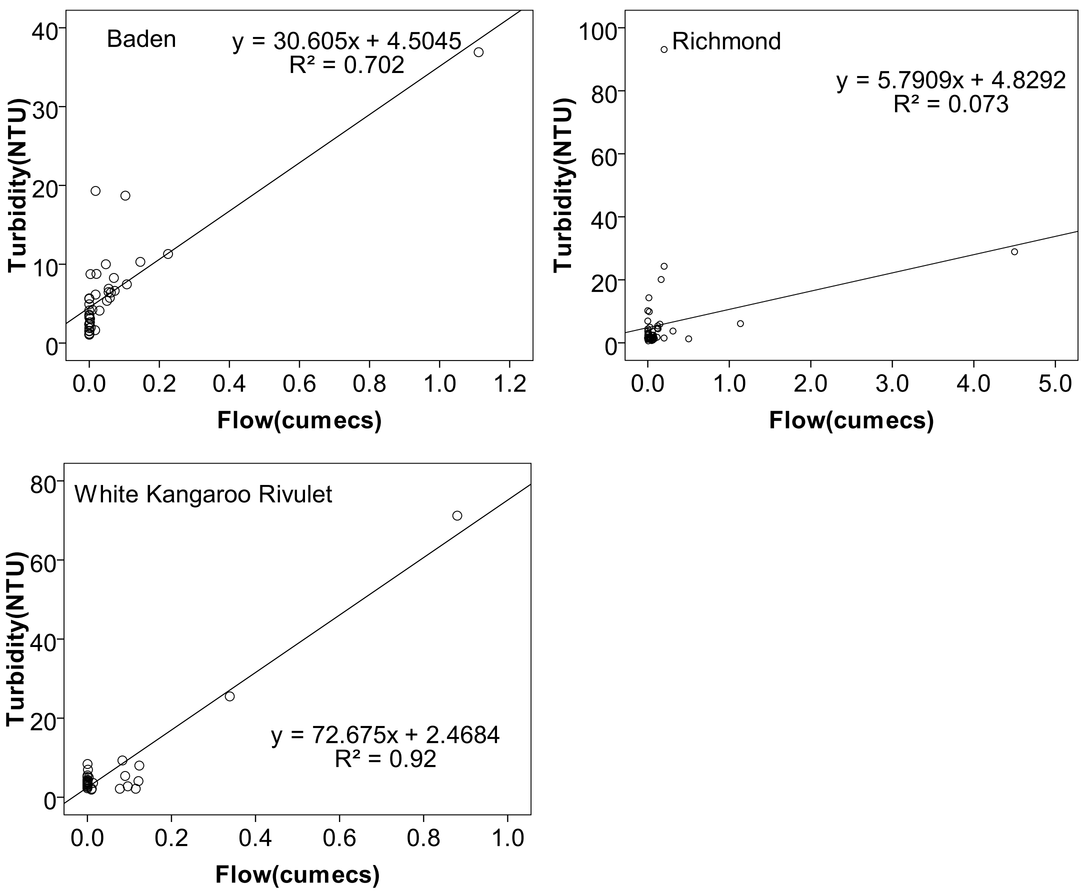


Figure 49 Significant relationships between turbidity and stream flow at Baden and Richmond in the Coal River and White Kangaroo Rivulet.

The poor relationship at Richmond station may be the effect of riparian management activities of Coal Valley Landcare group that includes fencing off paddocks from the river and planting of native trees on the riparian strip. The poor relationship at Richmond seems to be mainly caused by a single very high turbidity event at low flows. While the riparian management activities may play a role, site disturbance during a period of low flow seems to have affected the point. High flow can cause erosion of river banks and carry sediment particles. High flow may be due to heavy rainfall, storms or water input from drainage lines. Bolstad & Swank (1997) found that turbidity reached its maximum during storms in the Coweeta Creek Watershed in Western North Carolina. Storm water that causes high flow is responsible for exporting a high percentage of catchment pollutants including nutrients and sediments from their source of origin like agricultural fields (Drewry *et al*. 2006) through gully and bank erosion (Smith *et al.* 2005).

The regression analysis showed that stream flow was poorly related to nutrient concentrations at Baden, Richmond and Downstream of Craigbourne Dam but there was significant relationship ($p<0.05$) at White Kangaroo Rivulet (Figure 50). This indicates that nutrient concentrations (especially phosphorus) in river water were dependent upon the amount of stream discharge in White Kangaroo Rivulet gauging station. Similar findings were reported by Reuss *et al.* (1997), Swank *et al.* (2001) and Wang *et al.* (2006) in subalpine forest at Fraser in Colorado, southern Alpplachain catchment in North Carolina, and the Catskill Mountains in New York, respectively. Novak *et al.* (2003) showed a significant positive linear relationship between $\log_{10}$ stream flow and $\log_{10}$ dissolved phosphorus in the agriculturally intensive South-east Coastal Plain watershed in North Carolina, United States of America indicating that high summer flow increased dissolved phosphorus export to the river.

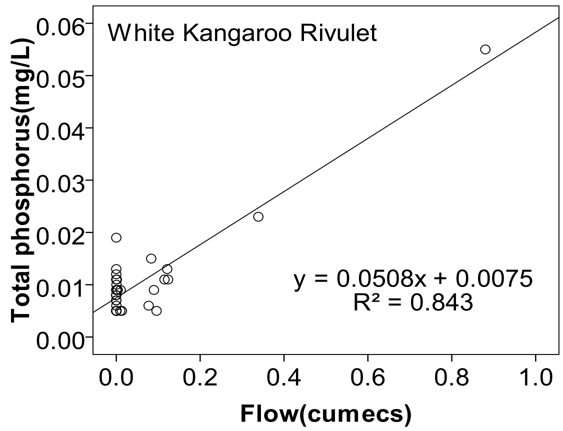


Figure 50 Significant relationship between river flow and total phosphorus concentrations at White Kangaroo Rivulet gauging station.

Similarly, other studies in the Southeast Coastal Plain watershed showed increased P export during high stream flow (Asmussen *et al.* 1979; Lowrance & Leonard 1988). This was due to flood waters removing sediment-bound phosphorus from riparian areas (Poinke *et al.* 1999; McDowell *et al.* 2001). Gökbulak *et al.* (2008) reported a significant positive correlation between stream discharge and nutrient losses in an Oak-Beech forest watershed in Istanbul, Turkey. Verhoff *et al.* (1982) showed the total phosphorus (TP) concentration depends on water flow in the rivers of Western Ohio where peak TP concentration coincided with the peak flow in the river. Conversely, Costello *et al.* (2000) reported that phosphates tend to show slight seasonal variability in the river water although concentration may increase a little during periods of the low flow/ runoff.

The strongest correlations were between turbidity and nutrients, particularly TP in the Coal River. Stream flow and turbidity had the highest correlation of all data, but only for the White Kangaroo rivulet. Total phosphorus and flow had a very strong correlation in the White Kangaroo but not total nitrogen suggesting the nitrogen may be in part sourced from groundwater. This is supported by the high nitrate levels in White Kangaroo Rivulet.

5. 3. Land use and water quality interactions

Land use mapping was generated from 2005/7 aerial photographs. Land use classes were digitised for the riparian zone which for the purposes of this study was defined as one kilometre either side of the Coal River and White Kangaroo Rivulet. Others small Creeks and Rivulets were not studied as there was no water monitoring stations with which to make comparisons.

5. 3. 1 Whole catchment riparian land use

Land use in the study area within one kilometre of the river showed that native forest and native pasture were the major land cover/land use types followed by improved pastures. Figure 51 and 55 show the major land cover in the riparian zone in the Coal River and White Kangaroo Rivulet were native forest (33.7%) and native pasture (27.7%) followed by the improved pasture (17.3%).

Table 6 Riparian land use within one kilometre buffer in the Coal River and White Kangaroo Rivulet.

Land use categories	**Year 2005/6**	
	Area(km^2)	%
Cereal Cropping	11.92	6.43
Dam-lake	1.95	1.05
Home paddock	2.70	1.45
Improved pasture	32.05	17.29
Intensive horticulture	0.70	0.38
Native forest	62.47	33.70
Native pasture	51.34	27.69
Pasture + trees/shrubs	14.98	8.08
Perennial horticulture	2.81	1.52
Plantation forest	1.56	0.84
Residential area	1.15	0.62
Willow riparian trees	1.78	0.96
Total	185.39	100

5. 3. 2 Baden subcatchment riparian land use

The main land uses in the Baden subcatchment was improved pasture (47%) followed by native forest (15.7 %), cereal cropping (15.46 %) and native pasture as shown in Figure 55 and Table 7. Figure 52 shows a small area of plantation forest at the head waters of the river.

Table 7 Riparian land use within one kilometre buffer in the Baden subcatchment in the Coal River valley.

Land use categories	**Year 2005/6**	
	Area(km^2)	%
Cereal Cropping	2.26	15.46
Dam-lake	0	0
Home paddock	0.33	2.27
Improved pasture	6.89	47.08
Intensive horticulture	0	0
Native forest	2.3	15.74
Native pasture	1.75	11.98
Pasture + trees/shrubs	0.92	6.26
Perennial horticulture	0	0
Plantation forest	0.18	1.22
Residential area	0	0
Willow riparian trees	0	0
Total	14.63	100

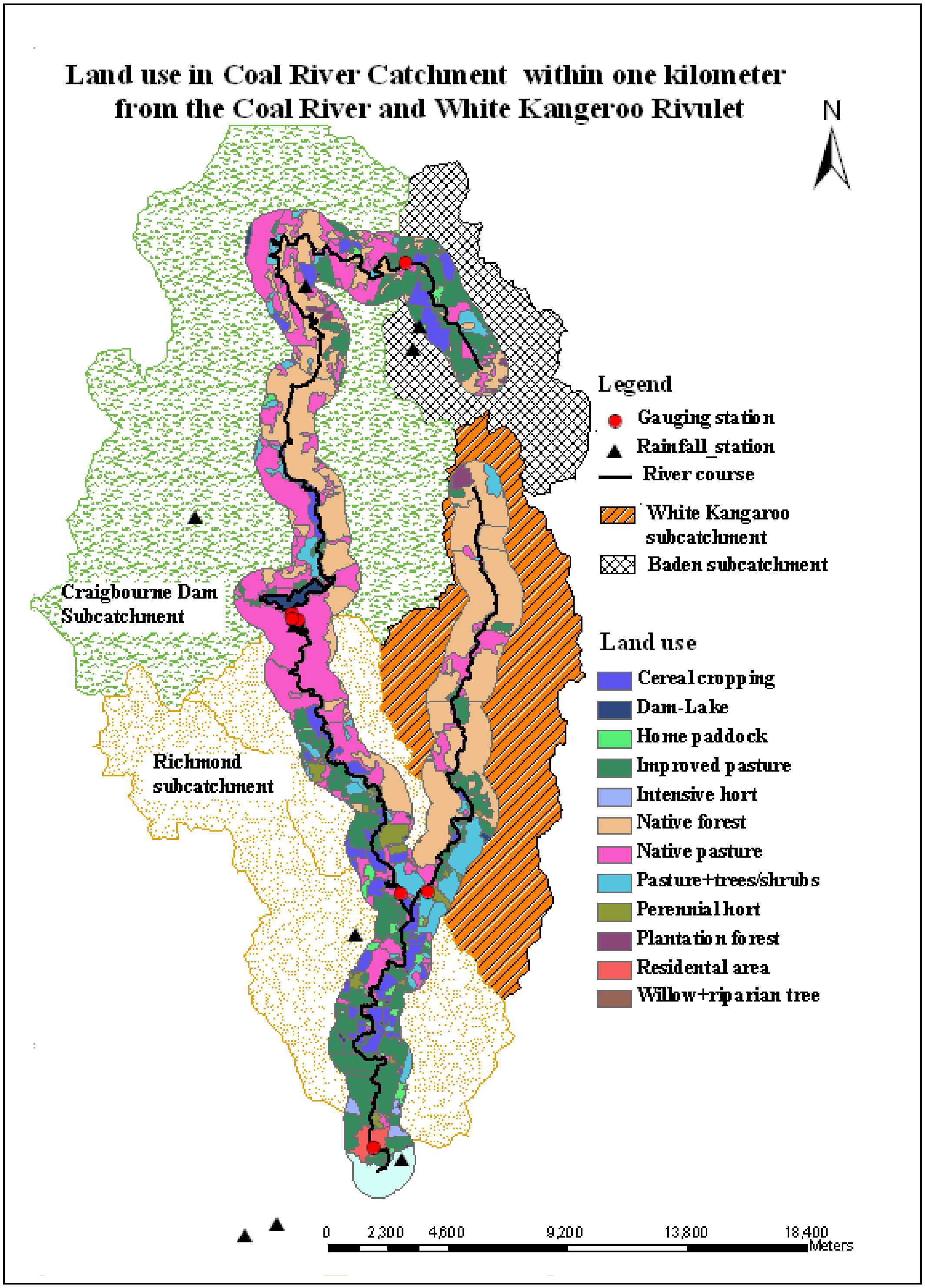


Figure 51 Land use within one kilometre of the Coal River and White Kangaroo Rivulet in Coal River Valley showing four subcatchments based on the location of gauging stations.

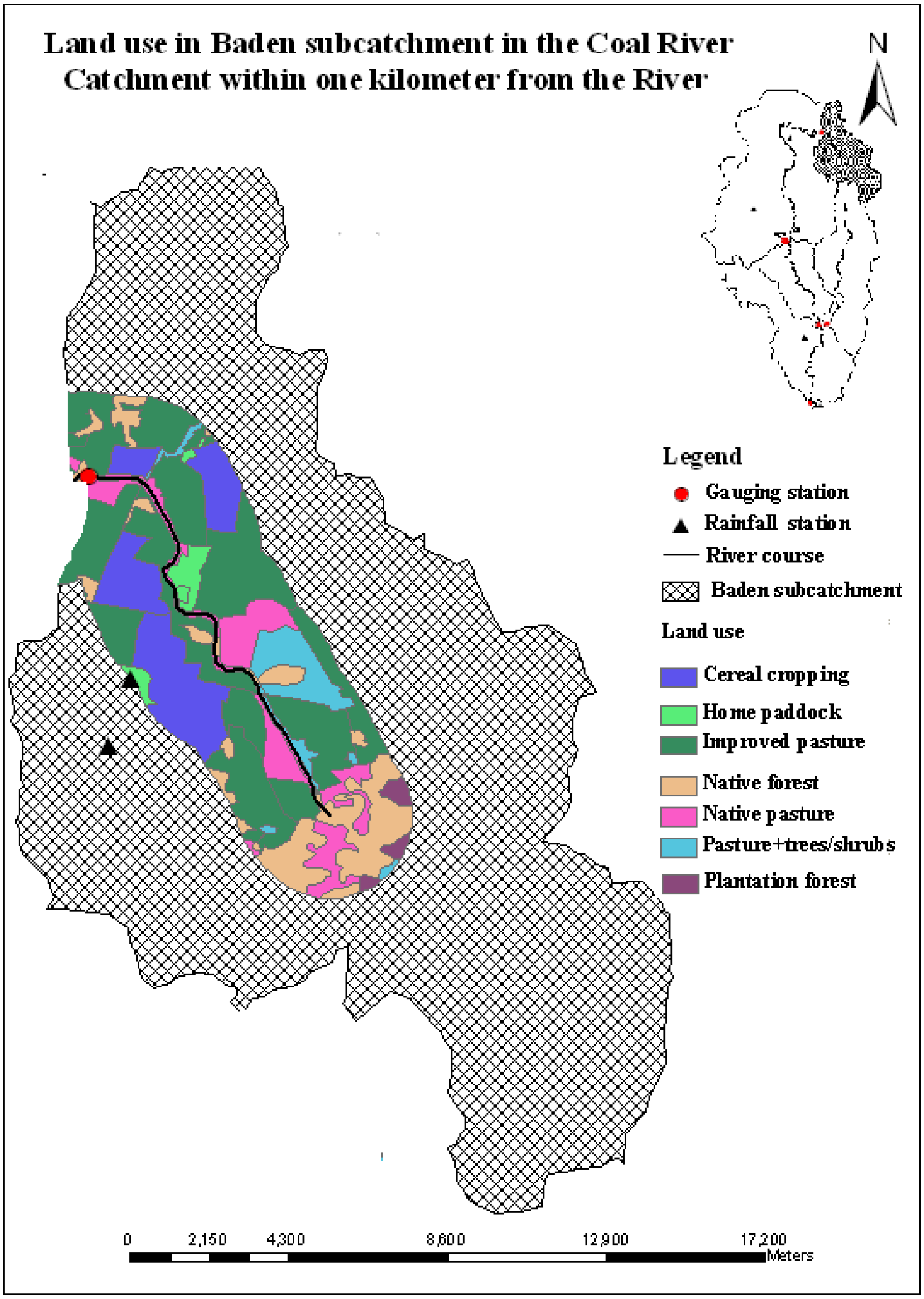


Figure 52 Land use share within one kilometre from the Coal River in Baden subcatchment.

5. 3. 3 Craigbourne Dam subcatchment riparian land use

In this subcatchment native pasture (42.74%) and native forest (37.51%) were the dominant land use type followed by the improved pasture (Figure 53 and 55). The area and percentage of the land use is given in the Table 8.

Table 8 Riparian land use within one kilometre of the river in Craigbourne subcatchment in the Coal River valley.

Land use categories in	**Year 2005/6**	
	Area(km^2)	%
Cereal Cropping	1.72	2.76
Dam-lake	1.47	2.35
Home paddock	0.27	0.43
Improved pasture	5.11	8.19
Intensive horticulture	0	0
Native forest	23.42	37.51
Native pasture	26.68	42.74
Pasture + trees/shrubs	2.54	4.07
Perennial horticulture	0	0
Plantation forest	0.82	1.31
Residential area	0	0
Willow riparian trees	0.4	0.64
Total	62.43	100

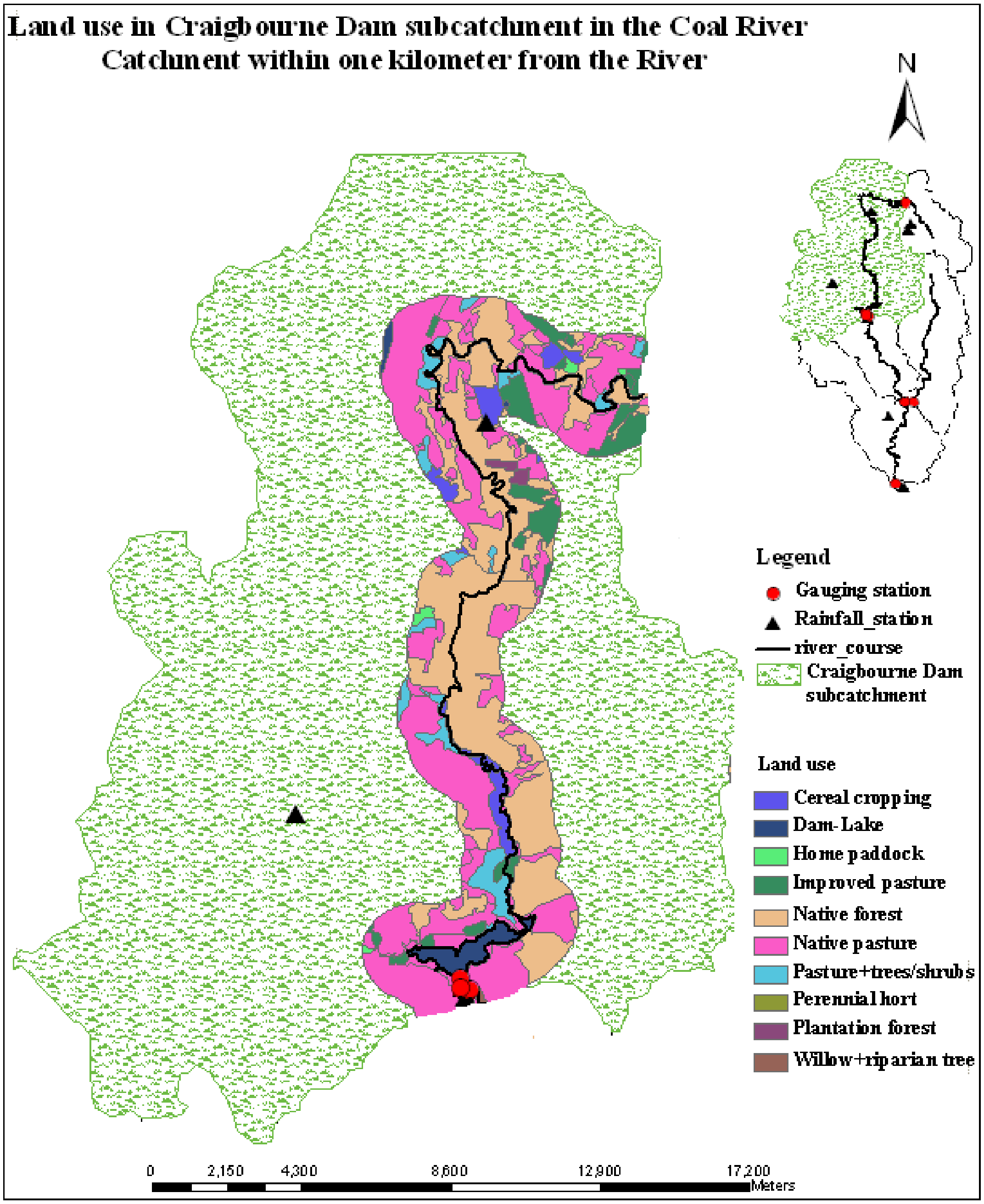


Figure 53 Land use type in Craigbourne Dam subcatchment in the Coal River Valley.

5. 3. 4 Richmond subcatchment riparian land use

Figure 55 shows land use in the Richmond subcatchment where the dominant land uses were native pasture and improved pasture. The main land use area and percentage is given Table 9. Figure 54 shows the spatial distribution of land use type with intensive horticulture, perennial horticulture and residential areas immediately upstream of the gauging station.

Table 9 Riparian land use within one kilometre of the river in the Richmond subcatchment in the Coal River valley.

Land use categories	**Year 2005/6**	
	Area(km^2)	%
Cereal Cropping	7.75	11.68
Dam-lake	0.37	0.55
Home paddock	2.08	3.13
Improved pasture	16.98	25.60
Intensive horticulture	0.70	1.05
Native forest	9.05	13.64
Native pasture	19.26	29.03
Pasture + trees/shrubs	5.45	8.21
Perennial horticulture	2.30	3.47
Plantation forest	0	0
Residential area	1.15	1.73
Willow riparian trees	1.27	1.92
Total	66.34	100

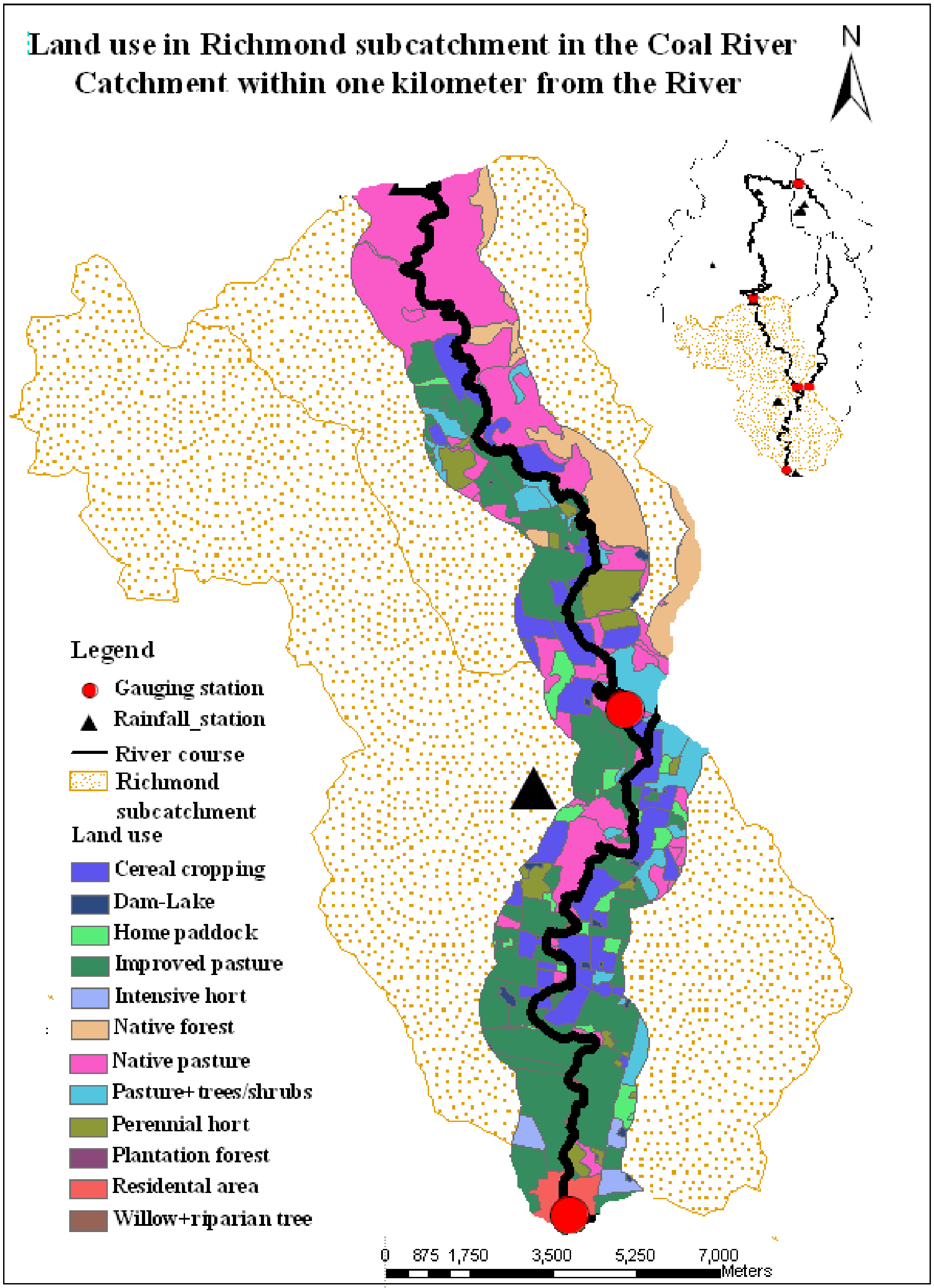


Figure 54 Land uses in the riparian zone of the Richmond subcatchment in the Coal River valley.

5. 5. 5 White Kangaroo subcatchment riparian land use

Figure 55 shows the native forest was the main land use type followed by the pasture + tree /shrubs in the White Kangaroo subcatchment. At the top of the White Kangaroo Rivulet plantation forests were grown and improved pasture was dominant near the riparian areas (Figure 56).

Table 10 Riparian land use within one kilometre of the river in the White Kangaroo Rivulet subcatchment in the Coal River valley.

Land use categories	**Year 2005/6**	
	Area(km^2)	**%**
Cereal Cropping	0.19	0.45
Dam-lake	0.11	0.26
Home paddock	0.02	0.05
Improved pasture	3.07	7.31
Intensive horticulture	0	0
Native forest	27.7	65.97
Native pasture	3.65	8.69
Pasture + trees/shrubs	6.07	14.46
Perennial horticulture	0.51	1.21
Plantation forest	0.56	1.33
Residential area	0	0
Willow riparian trees	0.11	0.26
Total	41.99	100

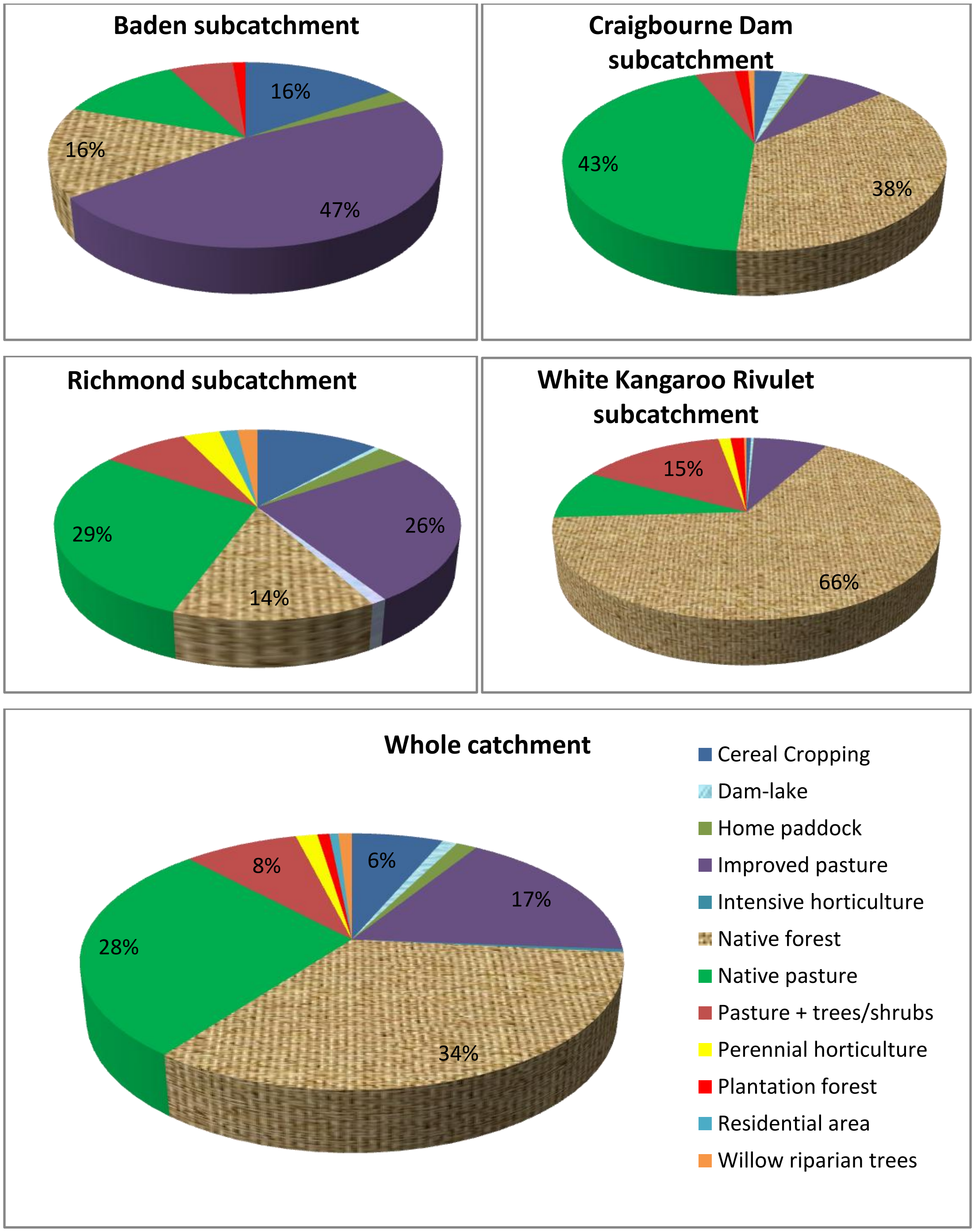


Figure 55 Land use share within one kilometre from the Coal River and White Kangaroo Rivulet in the Coal River valley.

In summary there are very distinct differences in subcatchment land use cover. The Baden subcatchment is most dominated by improved pastures while Craigbourne is a mix of native pasture and native forest. White Kangaroo is dominated by native forest and Richmond is mixed but with the most perennial horticulture and riparian willows.

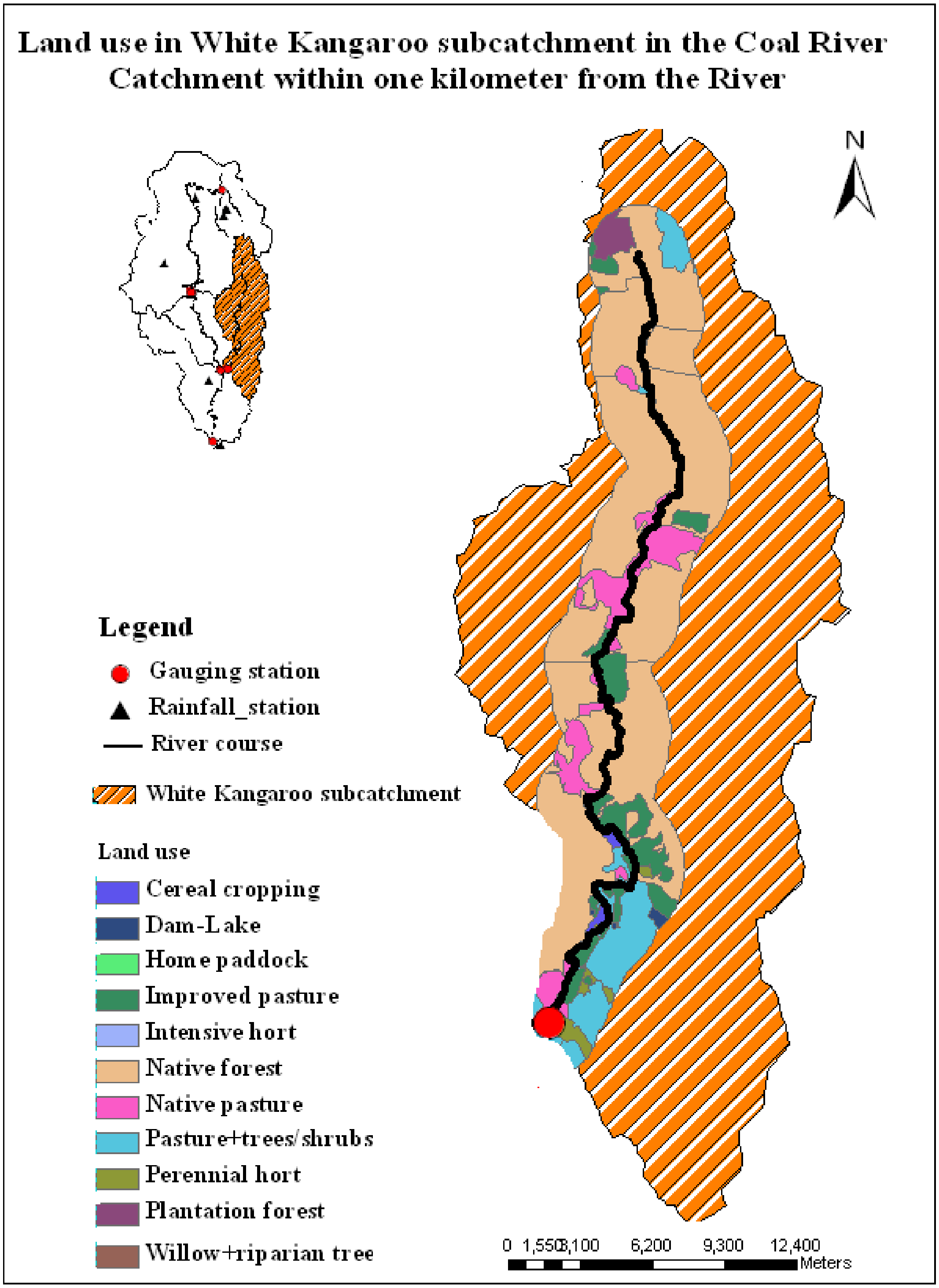


Figure 56 Land use type in White Kangaroo subcatchment in the Coal River valley.

5. 5. 6 Riparian works in the Coal River banks in the Richmond subcatchment

Coal Valley Landcare Group has carried out significant management of riparian vegetation and related work from below Craigbourne Dam to Pitt Water at the mouth of the Coal River. When the willows choking the river channel and immediate flood plain were associated creation of a new river channel at “Riversdale” on the Colebrook Road during a flood in 1963, residents of the Coal River valley made efforts to manage the river system. They undertook willow removal activities and constructed a diversion bank across the top of the erosion gully generated by the 1963 flood (Lisson *et al.* 1997). The physical impact of willows on river beds leads to the blockage of the river course and eventual diversion of the channel (Lester *et al.* 1994). During the removal programme many willow trees were replace by the pastures grass or native trees.

In 1990, large scale willow removal programmes were initiated in the Coal River system funded by Landcare and the Natural Heritage Trust (Bobbi 1999) with the aim of waterway rehabilitation because willows were widespread along the course of the river and blocked the river channel as shown in Figure 9. The choking effect of these trees led to the severe flood in 1963 at Riversdale and Colebrook. The waterways rehabilitation activities included riparian willow removal, stabilisation of the river bank by planting native plants and fencing river bank to prevent stock access to the river. However the affects of this activity on water flow, bank erosion and water quality is not known. In 1993 approximately 6,298 m of willow infested stream length was cleared on the Coal River at Stockdale near Campania, followed by a further 478.5 m in 1994 with the stumps painted with herbicide immediately after cutting (Mendham 2002b). Between April and July of 1999, willows were also removed from approximately 5,598 m of river banks from Eliza Farm and Campania House. Between April 1998 and December 2000 a total 4 km of previously cleared riparian land was fenced and revegetation work completed (Mendham 2002a).

In 2000 in the Rosedale-Stockdale and Colebrook Dale- Barton Vale area, a total of 2,416 m of willows were removed, followed by a further 381 m in 2001. The cleared area of river bank was either planted with native species or left to encourage the natural regeneration of native species (Mendham 2002a). In 2002, 4,095 m of river bank near Richmond and was cleared of willows and 5 km of fencing was completed with 2,500 native trees planted to prevent the river bank erosion. During 2005 willows were cleared from approximately 4,636 m along the Coal River bank with 4 km fencing and 2,600 native trees were planted (Mendham, 2005). In 2007 a further 530 m of willows were removed from river bank.

In total, some 24.43 km of willows have been removed from the banks of the Coal River between downstream of Craigbourne Dam and Richmond over a 14 year period. In addition, 13 km river banks have been fenced to manage stock access to the river (Figure 57).

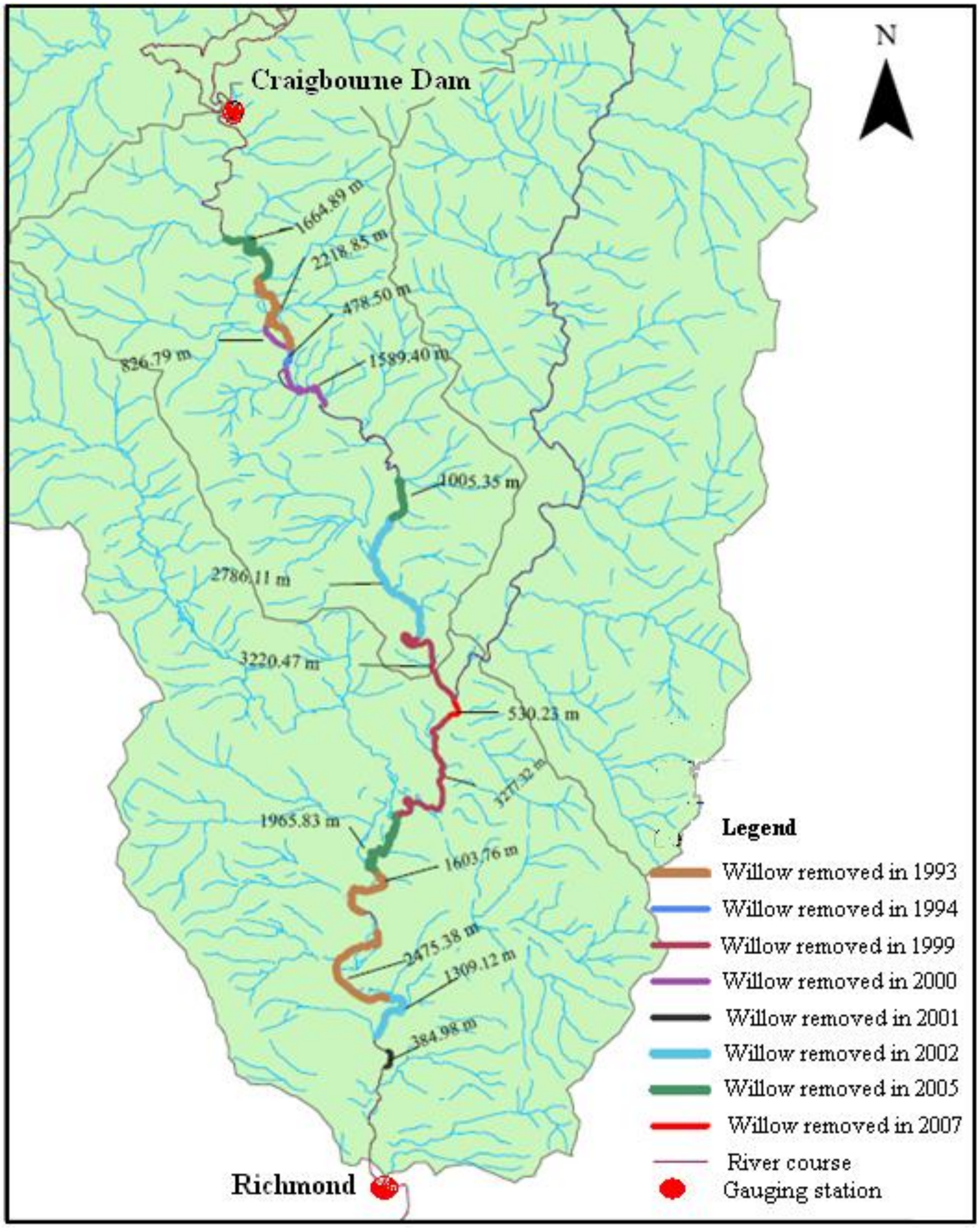


Figure 57 Willow removal programmes below the Craigbourne Dam to Richmond between 1993 and 2007 in the Coal River valley.

5. 5. 7 Linkage between land use and water quality

In order to examine any links between land use and water quality, the land use mapping derived from 2005/7 aerial photograph was compared to the 2005/7 DPIPWE water quality data for the four gauging stations in the catchment.

a) Water Temperature

Figure 58 shows slightly higher mean temperature at the Richmond station (13.4 ºC) but there were no significant differences between water temperatures at the four stations ($p<0.05$).

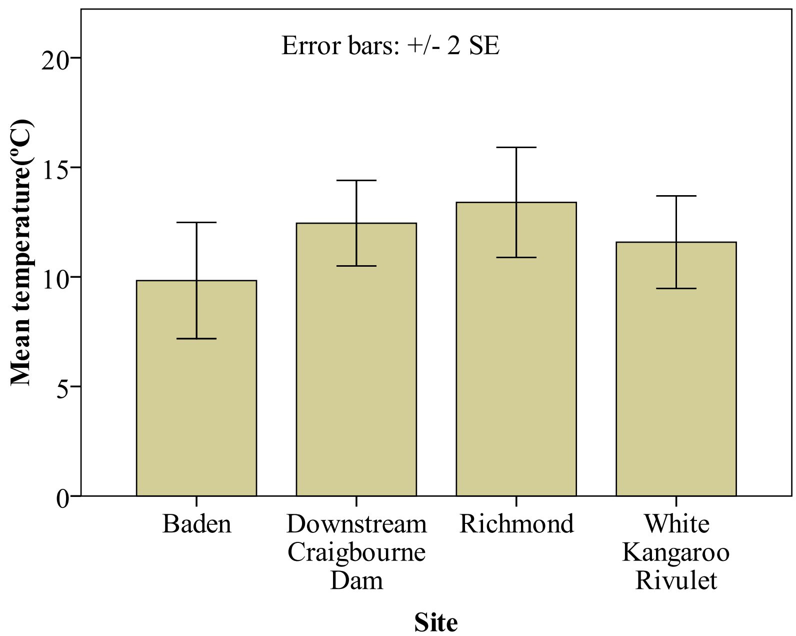


Figure 58 Mean water temperature at four gauging stations in the Coal Valley for 2005/7.

Spatial variation in temperature both within and between streams has been linked to several natural and anthropogenic factors. In spatial terms stream temperatures differ due to elevations, aspect, rainfall, groundwater recharge and riparian land use. Higher stream water temperature may be the effect of land use in the Richmond subcatchment where riparian land use has 20% residential areas (Figure 55). Due to increased impervious surface cover during urbanization accumulation and drainage of rainwater from hot paved surfaces (such as roads and paved areas) help to increase the stream temperature

(Pluhowski 1970). However, a study conducted in an urban stream in central Tokyo showed that a major cause of increases in the water temperature was input of heat from wastewater (Kinouchi *et al*. 2007). LeBlanc *et al.* (1997) reported that shade of riparian trees, groundwater discharge, and stream width had significant effect on the stream temperature. Anbumozhi *et al.* (2005) reported that riparian buffer provides shading effect to lower stream temperature. Figure 56 shows that approximately 4.6 km of the Coal River banks were cleared of willows in Richmond subcatchment in 2005. So, cleared river banks reduce shading on river that may be the cause of increased temperature. This is supported by the findings of others researcher reported where daily mean water temperature was increased by as much as 2 to 10 ^{0}C when trees were harvested on the river banks (Brown & Krygier 1970; Martin *et al*. 1985; Stott & Marks 2000). In addition, many researchers suggested that riparian plants control and regulate the river thermal properties by capturing short wave radiation during the day time and preventing heat loss from long wave radiation at night (LeBlanc *et al.* 1997). Metzeling (1977) reported that cleared sections of stream bank of the Yarra River catchment showed higher temperature ranges than partly cleared or well vegetated sections of the same streams. The most important harmful effect of the willows on the riparian zones is the blocking of the sun light which has a significant impact on the water temperature and population of the stream flora and fauna (Dawson & Haslam 1983). In general if the river bank is not covered by plants more light penetrates the river which ultimately increases the water temperature in the day time.

Similarly, summer maximum river temperatures in the Toikanbetsu River basin of northern Japan has increased from 22 °C in 1947 to 28 °C in 1989 due to continuing damage of the riparian forest (Nagasaka & Nakamura 1999). Fluctuated and higher temperature has been recorded on the river bank without riparian vegetation shading (Quinn *et al.* 1992; Harding & Winterbourne 1995; Quinn *et al.* 1997). In addition, during warm weather high solar intensity on the wide, broad and shallow river channels are more likely to heat up. This may be one reason why higher temperatures occur at the Richmond station. Pluhowski (1970) reported that water temperature on Long Island urban streams showed mean summer temperature warmer by 5–8 ºC and cooler approximately 1.5–3 ºC in winter than the forested streams. This was the effect of removal of riparian vegetation, decreased groundwater recharge, increased impervious surface affects on the stream temperature.

In addition, the river flow at Richmond is less than that downstream of Craigbourne Dam station even though there is additional flow from the White Kangaroo Rivulet. So less water and wider and winding river paths may help to increase the river temperature at the Richmond station. On the other hand, residential waste water and storm water may have influence to increase the water temperature. Similarly, the urbanization/residential infrastructure has an effect on reduction on the base flow due to increased developments of impervious surface and reduce the infiltration potential of rain and reduce ground water recharge during rainfall. As a result, low base flow may have effect on the low volume of water flow in stream (Leblanc *et al.* 1997). However, stream temperature is not only influenced by upstream land use condition (Roth *et al*. 1996; Stott & Marks 2000) but also affected by wind speed, solar radiation, relative humidity, water depth, riparian vegetation (Pilgrim *et al.* 1998), stream size, stream surface turbulence and stream water travel time (Cluis 1972).

b) Dissolved Oxygen (DO)

Figure 59 shows that mean dissolved oxygen concentrations were significantly higher downstream of Craigbourne Dam (11.08 mg/L) than Baden (7.5 mg/L), Richmond (9.62 mg/L) and White Kangaroo Rivulet (9.22 mg/L) at $p<0.05$. Baden showed significantly lower DO than the other stations while there was no difference between Richmond and White Kangaroo DO values (Appendix 2.4). The higher nutrient concentrations in Craigbourne Dam are likely to have influenced the development of photosynthetic algae and higher levels of photosynthetic activity may have led to higher levels of oxygen production. Nutrients collected in dams from upstream land use activities such as cereals cropping, improved pasture and plantation forestry in the Baden and Criagbourne Dam subcatchments could have caused algae growth in the dam (Figures 52 and 53). In general, progressing upstream, the presence of DO in the river is lower as oxygen is more easily dissolved into water at low altitudes because of higher atmospheric pressure.

However, the DO downstream of Craigbourne Dam station is higher than the DO recorded in the Richmond gauging station, which may be explained by photosynthetic activity on the part of aquatic plants in the dam.

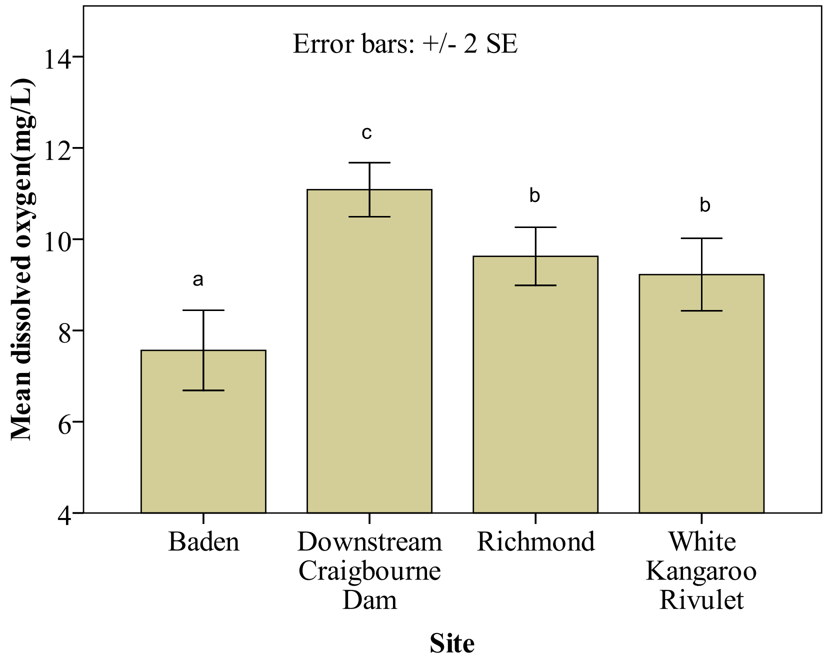


Figure 59 Mean concentration of dissolved oxygen (DO) at gauging stations in the Coal River and White Kangaroo Rivulet in 2005/7. Lower case letters denote results of Games-Howell test ($p<0.05$), used for a post hoc comparisons between means.

The others factor responsible for a significant reduction in the concentration of dissolved oxygen is human induced pollution. Biodegradable organic substances, excess waters from agriculture, waste water of the food and beverage industry and urban sewage that drain to the river are decomposed by certain species of aquatic bacteria and use dissolved oxygen for this process (Svobodova *et al.* 1993). Similarly, others mechanism that control the dissolved oxygen in the river system are solar radiation, photosynthesis and respiration. Daily fluctuation in dissolved oxygen in water is a common mechanism because during the day time aquatic plants produce oxygen by photosynthesis and at night-time they lose it via respiration process. During the day, photosynthetic activity is highest in the lower reaches of the basin due to channel widening and decreased riparian shading. The reason for there being higher dissolved oxygen in Craigbourne dam may be the effect of photosynthesis activities on the dam. It is noted that most of the data in this study were collected during business hours, i.e. in the day time. .

c) Turbidity

Figure 60 shows that mean turbidity was higher in the Baden subcatchment (4.70 NTU) followed by the White Kangaroo Rivulet (4.25 NTU) and Richmond (2.82 NTU). The lowest mean turbidity was observed at downstream Craigbourne Dam (2.37 NTU). High turbidity in Baden could be the effect of lack of riparian willow trees and a higher percentage of cereal cultivation (Figures 52 and 55). Similar results were observed by Cotching (2006) where water draining from cropped paddocks had high turbidity due to soil erosion. This was a major source of sediment load in rivers on the north–west coast of Tasmania (Cotching 2006). The lowest turbidity value is no doubt due to the effect of the Carigbourne Dam allowing sediments to settle out before measurement at the water monitoring station further downstream. However, the turbidity at Richmond is lower than the Baden and White Kangaroo Rivulets. This is likely to be the effect of higher percentage riparian vegetation cover such as willow trees and shrubs in the Richmond subcatchment as compared to the other subcatchments. Also, 4 km of riparian area was also fenced off during 2005/7 in the Richmond subcatchment. Similar result was observed in Buttons Creek in northwest Tasmania where turbidity level was reduced when the stream passed through native bush riparian area (Cotching & Sims 2003). Niaman & Decamps (1997) reported that 80 - 90% of sediments transported from the fields can be trapped in riparian zone with a diversity of vegetations. Similar findings were reported by Williamson *et al*. (1996) from New Zealand study where 85% of sediment loads were dropped when grazing was excluded from the river banks or erosion prone hills.

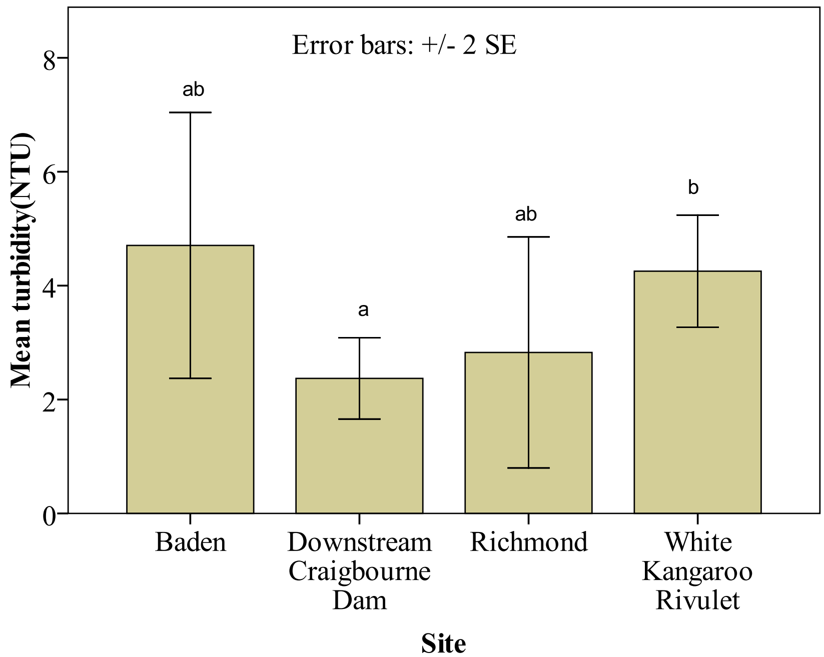


Figure 60 Mean turbidity at four stations in the Coal River and White Kangaroo Rivulet in 2005/7. Lower case letters denote results of Games-Howell test ($p<0.05$), used for a post hoc comparisons between means.

d) Electrical conductivity (EC)

Mean electrical conductivity (EC) values for 2005/7 are shown in Figure 61 which indicates values were higher in White Kangaroo station (1055.42 µS/cm) followed by the Richmond station (893.84 µS/cm), Baden(530 µS/cm) and lowest downstream of Craigbourne Dam (525.27 µS/cm). EC values from Richmond and White Kangaroo were significantly different from those of the other stations but not each other at $p<0.05$.

In general, there is a higher salinity, suspended solids and ammonium, in urban streams which can result from WWTP (waste water treatment plants) effluent, non–point source (NPS) runoff, illicit discharge connections, leaking sewer systems, and failing septic systems (Faulkner *et al*. 2000). The electrical conductivity showed a gradual increase from the upstream to downstream stations along the Coal River. These trends could be the attributed to the use of chemical fertilisers, low river flow, and effluent water discharges from the adjacent urban and suburban populations that are characteristic along the Coal River progressing downstream. Increase in land uses in the Richmond subcatchment such as residential and intensive horticulture may be the reason there is higher EC than the upstream. Additionally, the saline water mixed from White Kangaroo Rivulet may be another reason to be higher EC at Richmond than the upstream stations.

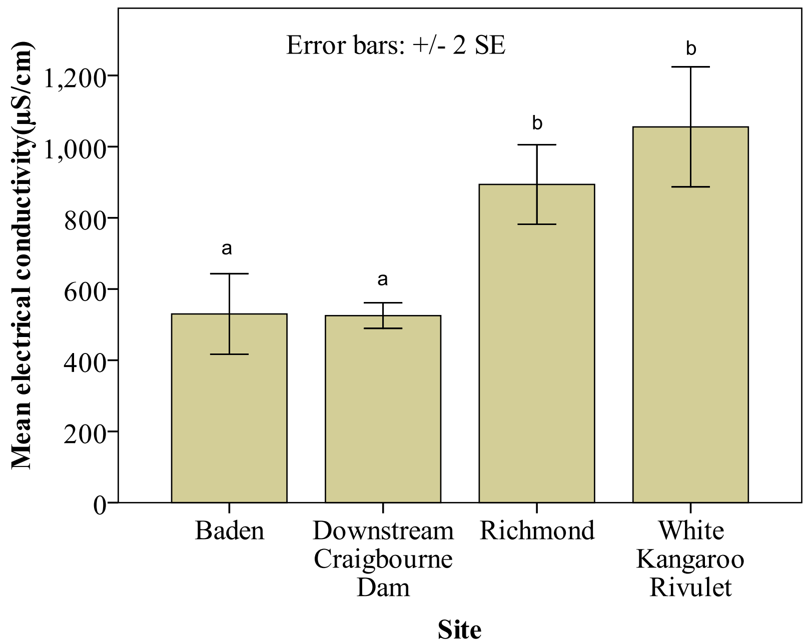


Figure 61 Mean electrical conductivity of different gauging stations during period of 2005/7. Lower case letters denote results of Games-Howell test ($p<0.05$), used for a post hoc comparisons between means.

In the White Kangaroo subcatchment, high conductivity may be the effect of geology of that area. Groundwater can become increasingly saline upon extended residence times in the certain geological materials. The local geological maps indicate the lithologies in White Kangaroo catchment are predominantly mafic dolerite on the western side and siliceous Triassic and Permian sedimentary rocks on the east on the valley (Figure 6). The stream follows along the geological contact. The two very diverse lithologies (mafic dolerite vs siliceous sediments) and their contact may cause groundwater to discharge into the stream. Bettenay (1978) and Engel *et al.* (1987) described the role of hydraulic barriers to groundwater flow causing saline groundwater upwelling, e.g., a dolerite dyke causing a saline seep in a groundwater system in Western Australia. Whether the saline groundwater is sourced from the dolerite or sedimentary rocks has not been established but it would be more likely sourced from weathered mafic rocks which have high nutrient levels and are more extensive in the White Kangaroo subcatchment. Also seeps are known to occur in the Tasmanian landscape where columnar rocks like dolerite or basalt contact horizontally bedded rocks like sandstones and mudstones (Rees 2000; Leaman 1973).

e) Water pH

The mean water pH value in the Coal River was higher at Craigbourne Dam station (8.13) which is probably the effects the dam discussed earlier. While the river water passing through the dam is mixed with the water from White Kangaroo Rivulet and diluted so it had slightly lower pH values than the dam but higher than the Baden station. Lower pH in the Baden may be due to the higher chemical fertiliser application on the improved pasture and cereal cropping which were the dominant land uses on that subcatchment (Table 7).

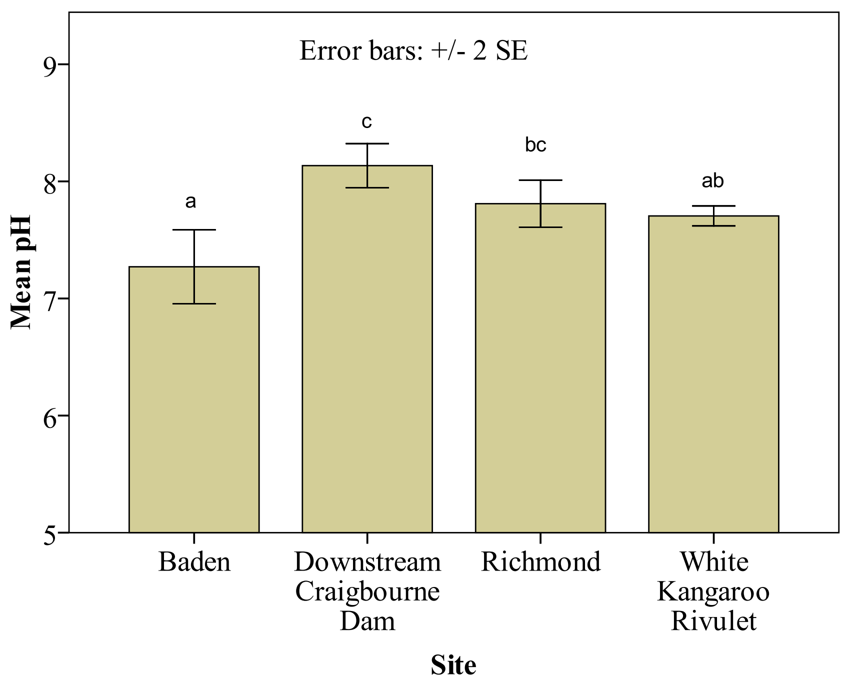


Figure 62 Mean water pH at four gauging stations on the Coal River and White Kangaroo Rivulet in 2005/7. Lower case letters denote results of Games-Howell test ($p<0.05$), used for a post hoc comparisons between means.

Most chemical fertilisers that are used in agricultural fields increase acidity by producing nitric acids (Helyar & Fenton 1999) although some types of manuring help to increase pH level (Whalen *et al.* 2000; Eghball 2002). However, Cotching (2006) reported that soil pH under cropping is higher compared to the pasture which may be due to the application of lime or dolomite. Another reason for the higher pH value in dam could be the result of excess photosynthesis activity due to the aquatic algae (Zweig *et al*. 1999).

f) Nutrients

1) Nitrate and total nitrogen

Figure 63 shows that mean concentration of nitrate was significantly higher at the White Kangaroo Rivulet gauging station (0.27 mg/L) than the other stations while those at Baden (0.004 mg/L), downstream Craigbourne Dam (0.020 mg/L) and Richmond (0.017 mg/L) were not significant different from each other at $p<0.05$ (Appendix 2.8). Total nitrogen values however were not significantly different between all four sites with downstream of Craigbourne Dam 0.71 mg/L, Baden 0.69 mg/L, Richmond 0.62 mg/L and White Kangaroo Rivulet station 0.63 mg/L.

This raises the question, what is driving the high nitrate values in White Kangaroo subcatchment? The land use in the White Kangaroo subcatchment was mainly natural forest, improved pasture and plantation forestry (Figures 55 and 56). Cotching (2006) suggests that leaching of applied fertilisers, mineralisation of organic matter or nitrogen fixation by the legumes may be an important source of nitrate in north-west Tasmanian rivers and creeks. One reason for high concentration of different forms of nitrogen could be the presence of high percentage of forest land with nitrogen fixing trees in the catchment. Also the ground water in the White Kangaroo Rivulet is highly saline and may also be a source of nitrate but this is unknown. High nitrate and salinity may well be a signal of groundwater influence. Davies (1988) reported a high percentage of nitrogen fixing plants such as *Acacia* and *Pultenaea* species in the forests of White Kangaroo subcatchment. Nitrification can be increased by the presence of the nitrogen fixing plant species such as *Accacia dealbata* and result in high total nitrogen and nitrate concentrations in soils (Ellis & Graley 1987). Dosskey & Bertsch (1994) reported that a riparian forest in a 12.6 km^2 watershed contributed 93% of total organic matter load exported annually into the river in South Carolina. These stored sources of organic carbon on the thick forest floor may be the source of nitrogen found in adjacent river water (Fenn & Poth 1999). Similarly, Vink *et al.* (2007) observed higher dissolved organic carbon (DOC) and nitrate (NO_3^-) concentration in the stream water draining from forested catchments in south eastern Australia. They suggest that higher NO_3^- in stream water due to increased nitrification as a result of breakdown of leaf litters. Another reason could be the formation of soils with high organic matter (high N and P) during decaying process of trees leaves, herbaceous litter, twigs and branches in forest land and transported into river water. Fenn & Poth (1999) have suggested that high nitrate export to rivers by old growth forest can be attributed to high leaching of nitrogen, because of their lower nitrogen demand and retention capacity but higher harvesting characteristics. The second highest mean concentration of nitrate and total nitrogen at Craigbourne Dam may be the effect of the higher percentage of native forest, lack of riparian vegetation on the river bank and accumulated effect of land use from the Baden and Craigbourne Dam subcatchment (Figure 55). Easy stock access to the river and large proportion of grazing land (native pasture) on the riparian areas could explain the high total nitrogen values in Craigbourne Dam.

Richmond had low total nitrogen concentrations despite having intensive horticulture and residential areas amongst the land use type. Land use data showed that Richmond subcatchment had high proportion of willows as compare to the other subcatchments (Figures 54 and 55). In addition, 5.1 km of the river banks were cleared of willow, fenced off and planted the native trees just upstream of the gauging station at the same time (in 2005 and 2007, Figure 57). Vought *et al*. (1995) reported that soil microbes found on the riparian forest and wetland can removed 100% of the nitrate present on this area by denitrification process. The literature suggests that the nutrient uptake by the actively growing plants or vegetation directly influence the supply of nutrients in water flowing through riparian areas (Dosskey *et al.* 2010) and the rate of the nutrient uptake depends on the age of the vegetation and stage of growth (Ericsson 1994; Ice & Binkley 2004).

The decomposition of plant debris collected from the agricultural fields and the riparian trees produces soil organic matter and humic substances that have large influence on chemical transformations and transport in soil (McFee & Kelly 1995). This organic matter has ion exchange capacity and can hold dissolved substance from percolating water by ionic attraction, hydrogen and ligand bonding even it presence is very small in the soil (Brady & Weil 2008).

Intensive land use such as intensive horticulture in the Richmond subcatchment where land managers use sprinkler irrigation that leads to the soil commonly being in a wet and at times saturated condition has a key impact on water quality in the riparian zones. In this condition, decomposition processes consume the limited supply of dissolved oxygen and as result soil microbes must search for the alternative electron acceptors such as nitrate, sulphate and oxidised iron to continue the further decomposition process (Hill 2000). When these compounds are transformed into chemically reduced forms such as iron to iron phosphate, nitrate to ammonium or nitrous oxide and nitrogen gas, their solubility and mobility in soil is also changed (Hill 2000; Duff & Triska 2000). So this may be one reason for having low concentration of nitrate in the Richmond subcatchment as compared to the Craigbourne Dam and White Kangaroo subcatchments. Another process of the removal of nutrients from the riparian zone is denitrification from the nitrogen enriched ground water. Hefting *et al.* (2005) reported that denitrification is the major source of nitrogen removal from the wetter soil than plant uptake in European riparian zones. Similarly, in studies from several Canadian riparian sites, Vidon & Hill (2002 a, b) reported that denitrification process removed nitrate at rates of 12 to 291 kg N/ha/year during high water table periods. During high DO conditions, decomposition of plant litter (organic matters) produces ammonium which is quickly nitrified and forms nitrate, and then denitrified to form nitrite and nitrous oxide gas (Clinton & Vose 2006).

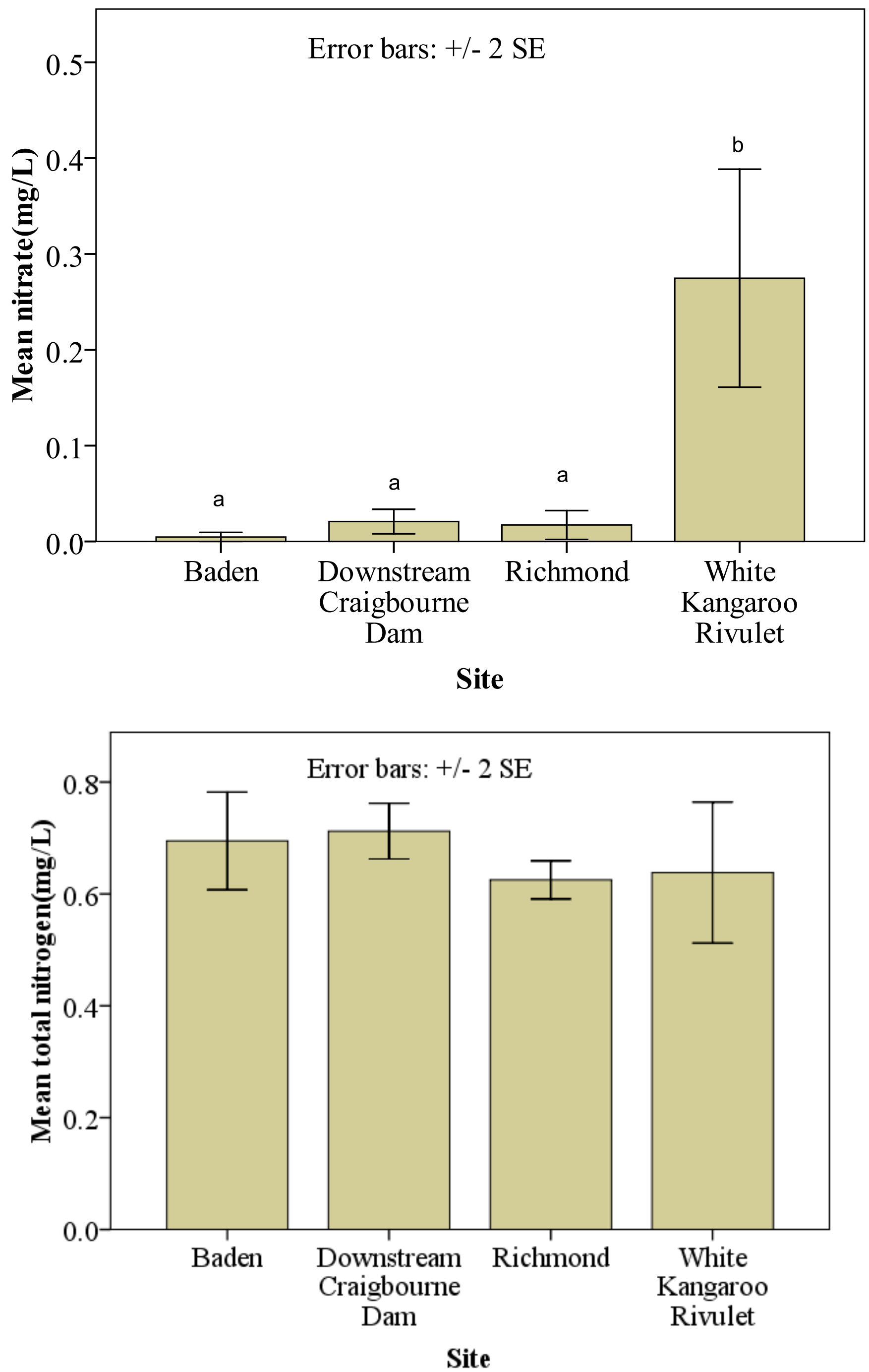


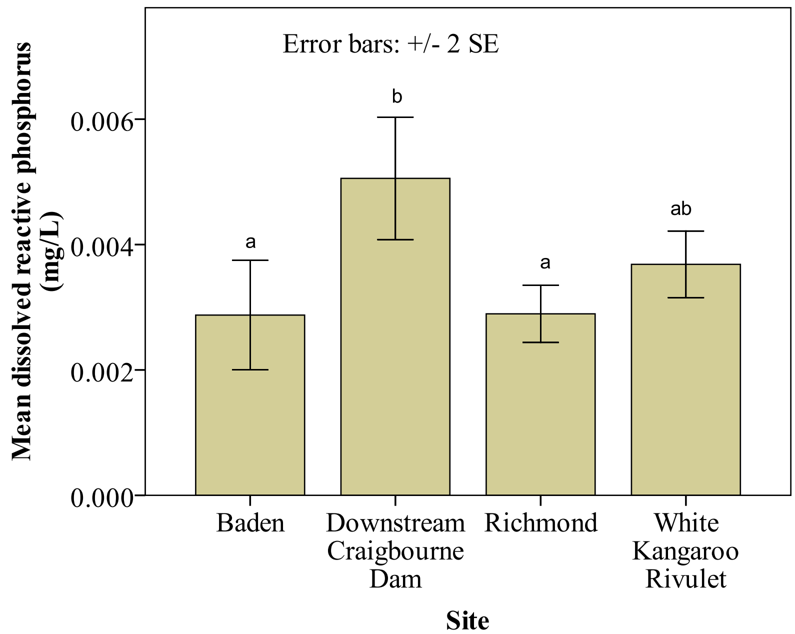
Figure 63 Mean concentrations of nitrate and total nitrogen at studied stations in the Coal River and White Kangaroo Rivulet in 2005/7. Lower case letters denote results of Games-Howell test ($p<0.05$), used for a post hoc comparisons between means.

Another reason for lower nutrient values on the Richmond site could be the restriction of livestock or sheep access to the river. In 2005 approximately 4 km of river were fenced off to avoid the livestock access to the river water and 2600 native trees were planted. This helps in the restoration of denuded and over grazed banks, livestock trampled riparian zones and lead to reduced catchment sediment export containing nutrients bound soil particles into the river. This suggestion is supported by lower mean turbidity values at the Richmond gauging station than Baden and White Kangaroo Rivulet in 2005/7 (Figure 60). Similar findings were reported by McKergow *et al.* (2003) from Western Australian Catchment where reduced catchment export of sediment from 100 kg/ha/year to less than 10 kg/ha/year were observed within one year after restoration of the denuded and overgrazed riparian zone. Clausen *et al.* (2000) reported that reduction of 35% nitrate concentration in groundwater and 83% and 73% in nitrate and total phosphorus concentration in overland flow through the riparian zone were observed within three years of restoration of riparian vegetation after long term annual crops cultivation in Connecticut, United States of America. So the effect of removal of willows and replacement with native riparian vegetation and fencing has been shown to reduce certain nutrients being exported to the river from the adjacent fields at Richmond station. Some researchers suggested that basin size also affects the nitrate concentration in the river. Wigington *et al*. (1998) described that concentration of nitrate decreased as basin size increased, presumably because of the nitrogen fixing alder trees influence in the smaller rivers.

Alexander *et al.* (2000) described that stream size affect the rate of loss of nitrogen from the water. And as water flow downstream nitrogen may be removed by biotic uptake or conversion to gas. Small streams with low flow (< 28 m^3/s) can lose half of their nitrogen compared to large stream with high flow. This may be a reason for low nitrogen concentration at Richmond as the Coal River channel widens downstream of Craigbourne Dam and stream flow diminishes. The lower concentration of nitrate at Richmond station is presumably because of a dilution effect as others small tributaries feeding into the river.

2) Dissolved reactive phosphorus (DRP) and total phosphorus (TP)

Figure 64 shows that slightly higher mean total phosphorus (0.22 mg/L) and dissolved reactive phosphorus (0.005 mg/L) were observed downstream of Craigbourne Dam. However, the only significant difference in phosphorus concentrations was the lower mean total phosphorus at White Kangaroo station than the rest of the stations at $p<0.05$ (see Appendix 2.2 and 2.7 for values). In addition, total phosphorus concentration of between 0.01-0.02 mg/L is considered typical for water draining from the intensive agricultural areas (Bobbi *et al.* 1996). While dissolved reactive phosphorus at White Kangaroo Rivulet was not statistically different than Baden, downstream of Craigbourne Dam and Richmond stations. Higher DRP values (0.005 mg/L) at Craigbourne Dam station may be the accumulated effect of Baden and Craigbourne dam. High amounts of phosphate in the dam may have the effect of increase in phosphate release from sediment under stagnant condition as reported by Van Vliet & Zwolsman (2008) in the Meuse River. Agriculture is the primary land use in the Baden, Craigbourne and Richmond subcatchments, and the use of phosphate fertilisers may be the reason for higher total phosphorus in those subcatchments compared to White Kangaroo station. In Baden improved pasture and cereal cropping were the dominant land use types within one kilometre of the river. Similar patterns were observed in total phosphorus concentration with higher values in the headwater sector where agriculture was the main land use type in the Pampean saline lowland Salado River, Argentina (Gabellone *et al.* 2005). Similarly, extractable phosphorus was found to be greater under cultivated land (particularly after potatoes) compared to long-term pasture on all soil classes in Tasmania (Cotching 2006). Storage of water by constructing the dam at the Craigbourne had significant effect on most of the water quality parameters. So, the increased nutrient concentration in the headwater regions could be due to use of fertiliser in agricultural activities which have been transported to the river through runoff.



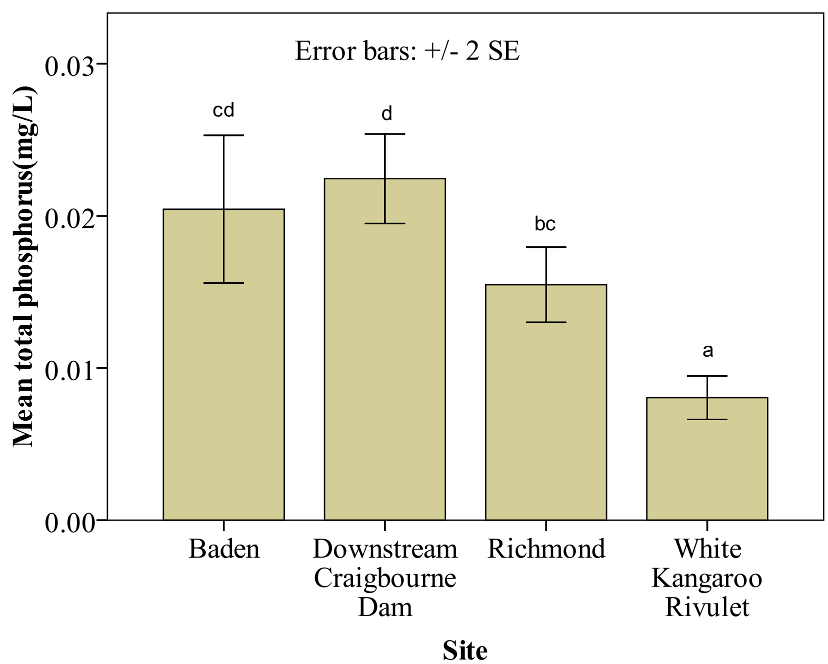


Figure 64 Mean concentrations of total phosphorus and dissolved reactive phosphorus at four gauging stations in the Coal River and White Kangaroo Rivulet in 2005/7. Lower case letters denote results of Games-Howell test ($p<0.05$), used for a post hoc comparisons between means.

On the other hand, in the Craigbourne Dam subcatchment native pasture is the dominant land use type which means a high probability of overgrazed and denuded hills. Also a good source of sediment movement in the river channel during rainfall and storm events can then collect in the Craigbourne Dam and become stored for long periods of time. Baden station (Figure 60) shows that turbidity was higher than rest of the observed stations which indicates a high amount of sediment was exported from the cultivated fields including improved pasture and cereal cropping areas into the river. A study of sedimentation into the Craigbourne Dam indicated annual influx of 3,463 tonnes/year, including 3.2 tonnes of phosphorus (Baker 2000). These nutrients bound to sediments moved to Craigbourne dam where they remained trapped for long periods of time. So the accumulated effect of land use of Baden subcatchment and Craigbourne Dam subcatchment could be one reason for higher concentration of dissolved reactive phosphorus and total phosphorus in Craigbourne Dam station. Little *et al.* (2003) reported that the variation in maximum total phosphorus concentrations in the Lower Little Bow River Watershed, Alberta, Canada could be explained by the proportion of cereal cropping in the sub-basin. A possible reason for lower concentration of phosphorus at Richmond than its upstream stations was that once water passed through the Craigbourne Dam, it was diluted by water with low total phosphorus concentrations from White Kangaroo Rivulet. Similarly, the Richmond subcatchment had higher willow riparian vegetation cover than the others subcatchment (Table 9) and some researchers reported that riparian buffers with grass strips can minimize phosphorus movement to the river (Lowrance *et al.* 1984; Novak *et al*. 2002). This may explain lower phosphorus concentration in Richmond than downstream of Craigbourne Dam. USGS (1999) reported that phosphorus and nitrogen concentrations were shown to be higher from the agricultural catchments than the urban catchments. However, the effect of residential areas and urbanization on higher concentration of total phosphorus had been reported by Meybeck (1998), Winger & Duthie (2000) and Osborne & Wiley (1988). Wastewater and fertilisers are one of the main sources of phosphorus in urban catchments (LaValle 1975). Cowen & Lee (1973) reported that trees leaves and seeds were the source for higher the DRP during the fall of the year.

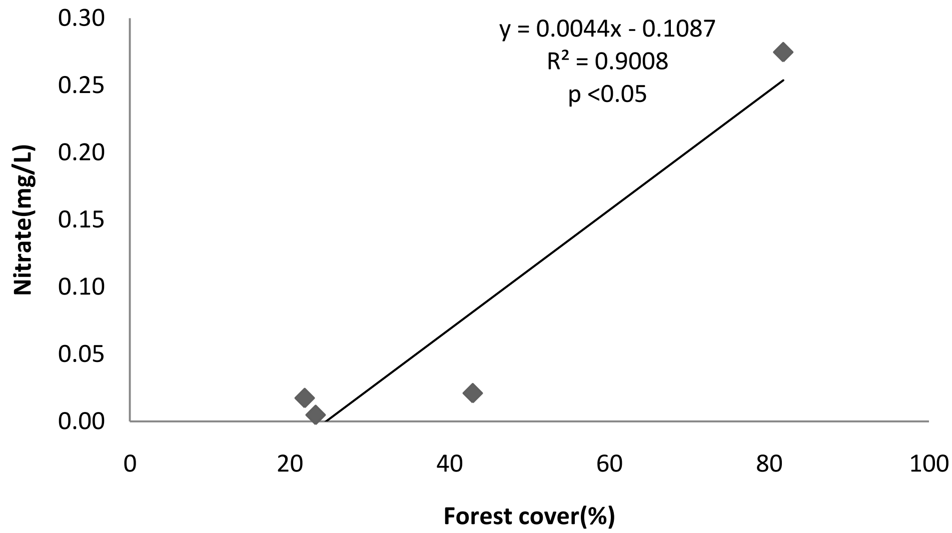
5. 5. 8 Relationship between land use and water quality

Demonstrating a clear link between land use and water quality is challenging as many factors can influence water quality and quantity. This is not surprising because important factors such as soil nutrient content, fertiliser application rate, cropping, irrigation methods, intensity of rainfall and intensity of crop management can have a major influence on river water quality. While the results of this study show significant correlations between two broad land use types, forest cover and native pasture cover, and nitrogen concentrations and turbidity respectively, these are based on very few sites and should therefore be treated with caution.

There was a significant positive correlation between forest cover (including native forest, plantation forest and pasture plus trees & shrubs) and nitrate (R^2 = 0.90). There was also a significant negative correlation between cover of native pasture and turbidity (R^2 = 0.90) No other land use types showed any significant relationships with water quality parameters at $p<0.05$ (Figure 65).

The observed positive relationship between proportion of forest cover and nitrate nitrogen concentrations contrast to the result of Meynendonckx *et al.* (2004) where they found nitrate concentrations were positively correlated with effluent loadings coming from wastewater treatment plant and agricultural land in River Scheldt basin, northwest Europe. These results suggest that forest cover could be a factor behind variation in nitrate nitrogen. However the results of regression analyses between land use percentage and water quality parameters cannot guarantee meaningful relationships with only four observation stations. The mechanism behind this observed relationship between native forest and nitrate nitrogen concentrations may be nitrogen fixation by leguminous trees and shrubs in native forests or some other natural process.

The significant negative correlation between native pasture and turbidity could be the effect of undisturbed land dominated by perennial plants which has the effect of reducing sediment loads to the river system. However, the relationships between land use and water quality parameters is very complex and correlations observed in any one watershed is likely to be site specific (Baker 2003) and is again dependant on very few sites.



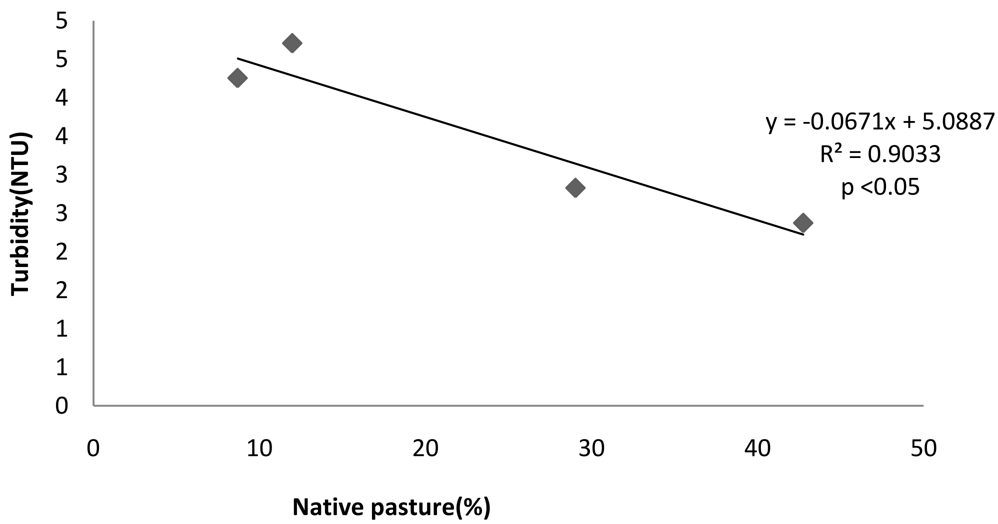


Figure 65 Significant relationship between land use percentage and river water quality in the Coal River valley.

5. 5. 9 Relationship between mapped geological units and water quality

The key geological materials in the catchment were spatially analysed to determine what role they may have in explaining variations in the water chemistry. Table 11 presents the percentage of the key (grouped) geological units in the various subcatchments as mapped at 1:250,000 scale by Mineral Resources Tasmania. The groups represent all the key rock types by geological ages and genesis e.g., igneous, sedimentary, metamorphic.

Table 11 Relationship between subcatchment geology and the water quality.

Subcatchment	Rock type	Area(km^2)	Percentage
Baden	Jurassic dolerite	20.13	38.24
	Permian sediments	24.65	46.82
	Quaternary deposits	0.97	1.84
	Triassic sediments	6.89	13.10
	Sub total	52.64	100
Craigbourne Dam	Jurassic dolerite	48.42	21.01
	Permian sediments	31.74	13.77
	Quaternary deposits	13.88	6.02
	Triassic sediments	126.19	54.75
	Tertiary basalt	0.51	0.22
	water	9.76	4.23
	Sub total	230.49	100
Richmond	Jurassic dolerite	63.27	38.83
	Permian sediments	0.35	0.22
	Quaternary deposits	0.01	8.87
	Triassic sediments	70.86	43.48
	Tertiary basalt	15.26	9.36
	Tertiary sediment	4.34	2.66
	Sub total	162.96	100
White Kangaroo	Jurassic dolerite	64.63	63.91
	Permian sediments	12.81	12.66
	Quaternary deposits	29.70	2.93
	Triassic sediments	20.72	20.49
	Sub total	101.13	100

Several interesting correlations were noted. While the high percentage of dolerite in the White Kangaroo was considered a possible cause of the higher salinity in that tributary the catchment wide correlation with salinity is not quite significant at $p(<0.05)$ (Figure 66). But lower pH of water in subcatchments showed strong trends to both lower percentages of Triassic sedimentary rocks (Figure 67) and higher percentages of Permian sedimentary rocks (not shown).

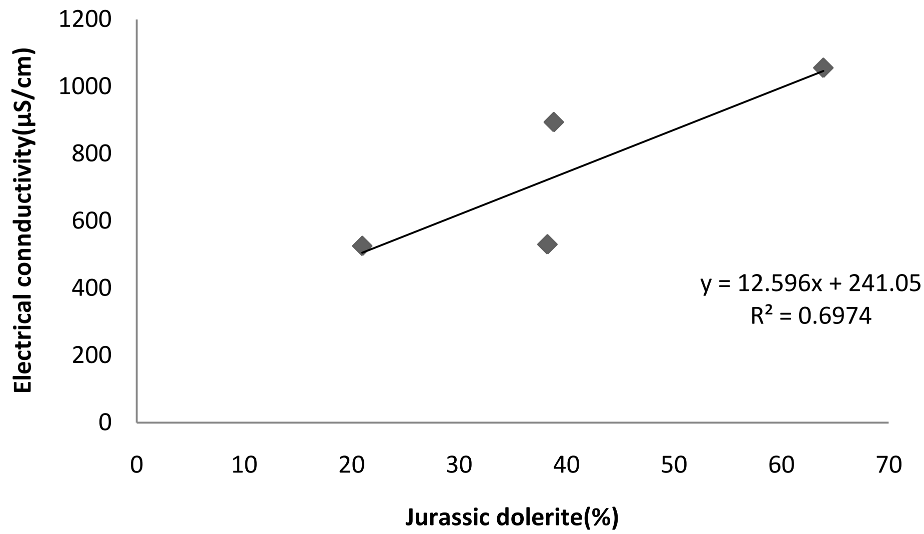


Figure 66 Relationship between electrical conductivity and percentage of Jurassic dolerite in the Coal River Valley. It shows the general trend of percentage of Jurassic dolerite and mean value of electrical conductivity on the river water for 2005/7 data.

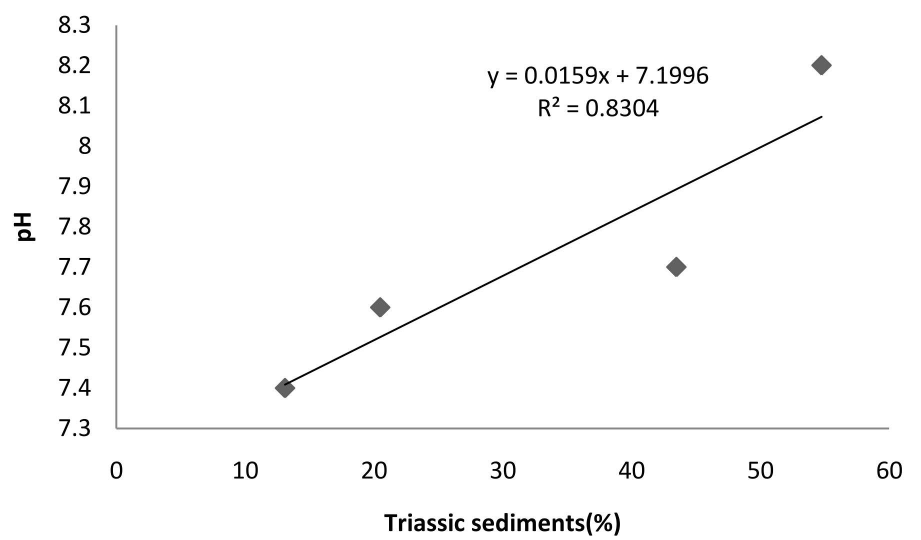


Figure 67 Relationship between pH and Triassic sedimentary rocks.

6. Conclusions

The Coal River Valley is considered a plank in the food bowl vision for Tasmanian agriculture. It is a demonstration of the impact of irrigation on land use change in Tasmania. In this study historical DPIPWE data were used to examine the spatial variations on water quality parameters in the Coal River. In addition, the linkage between riparian land use and selected water quality parameters were studied for the year of 2005/7. The following major findings were concluded from this study.

1. The historical data demonstrates that in-stream spatial variation of water quality for the rivers in Coal Valley catchment. There are no significant trends in temperature but higher values were recorded at the Richmond station. Similarly, electrical conductivity was higher at Richmond followed by White Kangaroo Rivulet then the downstream of Craigbourne Dam and Baden stations. The water quality parameters show complex geographical patterns. Mean water pH, dissolved oxygen (DO), dissolved reactive phosphorus (DRP), total phosphorus (TP) and flow show higher values at the station downstream of Craigbourne Dam. Nitrate nitrogen was higher at White Kangaroo Rivulet station but lower TP and total nitrogen (TN) were observed at Richmond station.
2. The impoundment of the river at Craigbourne Dam has shown a significant effect on the several water quality parameters such as pH, DO, DRP, TP, TN and flow. The higher values at the Craigbourne Dam may be the effect of chemical and biological interaction of aquatic organism with nutrient bound sediments trapped on the bottom of dam.
3. Dissolved oxygen is decreased with increasing water temperature and electrical conductivity in the Coal River. Turbidity and rainfall show significant positive correlations at all stations expect downstream of Craigbourne Dam. Stream nitrogen and phosphorus concentration is also positively correlated with the rainfall at Richmond.
4. Relationship of turbidity with nitrate was observed at Baden and Richmond stations and with total phosphorus at all stations but not at the station downstream Craigbourne Dam. Similarly, significant relationships between turbidity and total

nitrogen were seen downstream of Craigbourne Dam, Richmond and White Kangaroo Rivulet, but only Richmond station showed a positive correlation between turbidity and dissolved reactive phosphorus (DRP).

5. Stream turbidity was also significantly correlated with stream flow at Baden, Richmond and White Kangaroo Rivulet. Total phosphorus was correlated with the stream flow at White Kangaroo Rivulet station.

6. Riparian vegetation could be a strong determinant of reduced sediment load and thus regulate the nutrient concentration in the river system. In 2005/7 the Richmond station showed lower turbidity levels than the Baden and White Kangaroo Rivulet, this could be the impact of higher percentages of riparian vegetation such as willows.

7. The 2005/7 data shows riparian land management practices such as fencing and planting native vegetation on river banks also influence the nutrient concentration in the river by preventing the stock access to river and reducing sediment loads due to the sediment trapping role of vegetation.

8. Spatial variations in water quality are related with land use and natural factors. Results show mean nitrate was positively correlated with percentage of forest cover. This is most likely due to the role of nitrogen fixing plants in the native forest and the presence of high soil organic matter on the forest soil. Turbidity decreased with increasing percentage of native pasture in the riparian zone for 2005/7. Where native pasture provides an undisturbed soil surface.

Overall Baden is cooler and more acidic and has lower DO, DRP, EC and nitrate over both time series data sets. This seems to reflect its more elevated location in a more pristine part of the catchment. But it may also reflect the different geology and soils in that area where there is a predominance of Permian rock types.

Downstream of Craigbourne Dam has the most dissolved oxygen, highest water pH and is the least turbid. Lower turbidity probably relates to the still water while the higher oxygen could be due to either the daily biological activity or mixing during release from the reservoir. But the higher pH is likely to be due to the biological activity and water residence time in the reservoir. However it tends to have the highest total phosphorus and total nitrogen and this has been linked to sediment build up on the dam floor and to possible seepage of nutrients from the Colebrook sewage treatment plant. It has the highest flow and this relates to continuous water release from the dam.

The White Kangaroo subcatchment is the most saline (2005/7) and is much higher in nitrate. The nitrate may be due to the high acacia forest cover and the large percentage of dolerite bedrock. But the subcatchment has the lowest total phosphorus and this perhaps reflects the higher forest cover despite having higher water turbidity, which overall was strongly correlated to total phosphorus.

Richmond lies at the base of the catchment and is the warmest, most saline, over the longer term, and has the least total nitrogen (TN) and total phosphorus (TP) in the Coal River. Much of the salinity appears to be derived from the White Kangaroo tributary while the low TN and TP could relate to the higher percentage of riparian willows leading to the lowest turbidity, other than that of the Craigbourne reservoir.

This work has highlighted the need for long term water monitoring of key streams in Tasmania and the greater need to examine land use changes through time. This diverse catchment has shown several interesting features like the role of saline tributaries, the impact of reservoir storage on nutrients and turbidity and the impact of forest cover on nitrate levels. It has also supported the traditionally strong relationship between total P and turbidity.

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8. Appendices

Appendix 1.1 Summary statistics for 1999-2008 data, where 1= Baden, 2 = Downstream of Craigbourne Dam, 3 = Richmond, 4 = White Kangaroo Rivulet.

Water quality parameters	Site	Mean	S.D	S.E	95% Confidence Interval for Mean		Minimum	Minimum
					Lower Bound	Upper Bound		
Temperature	1	12.0961	5.95207	.83346	10.4220	13.7701	3.50	24.40
	2	13.3358	4.64112	.63751	12.0566	14.6151	4.60	21.40
	3	13.6327	5.12138	.69057	12.2482	15.0172	5.80	27.10
	4	12.1093	4.48049	.60972	10.8863	13.3322	3.30	24.10
Dissolved oxygen(DO)	1	8.0861	2.30053	.32214	7.4390	8.7331	3.60	14.90
	2	10.4783	1.29417	.17777	10.1216	10.8350	7.65	13.10
	3	8.8484	1.90323	.25663	8.3338	9.3629	4.23	14.70
	4	8.2246	2.77418	.37752	7.4674	8.9818	1.80	12.20
Turbidity	1	5.9520	5.92194	.82924	4.2864	7.6175	1.06	36.90
	2	3.4236	2.14740	.29497	2.8317	4.0155	1.08	9.04
	3	5.7564	13.24548	1.78602	2.1756	9.3371	.63	93.10
	4	7.6996	11.74789	1.59868	4.4931	10.9062	.93	71.20
Electrical conductivity (EC)	1	441.4157	190.00991	26.60671	387.9745	494.8568	196.20	1051.00
	2	564.0566	101.10132	13.88733	536.1896	591.9236	380.00	756.00
	3	859.5273	245.61554	33.11879	793.1281	925.9265	292.00	1327.00
	4	838.4815	373.32654	50.80331	736.5830	940.3800	315.00	1718.00
Water pH	1	7.4022	.53343	.07470	7.2521	7.5522	6.39	9.04
	2	8.2496	.39788	.05465	8.1400	8.3593	7.06	9.10
	3	7.7795	.32190	.04341	7.6924	7.8665	6.80	8.53
	4	7.6957	.19729	.02685	7.6419	7.7496	7.27	8.10
Nitrate	1	.006828	.0129727	.0024090	.001893	.011762	.0020	.0550
	2	.062192	.0795670	.0110340	.040041	.084344	.0020	.3630
	3	.033333	.0730769	.0099445	.013387	.053279	.0010	.4460
	4	.182491	.2115722	.0290617	.124174	.240807	.0020	.7570

Appendix 1.2 Summary statistics for 1999-2008 data, where 1 = Baden, 2 = Downstream of Craigbourne Dam, 3 = Richmond, 4 = White Kangaroo Rivulet.

Water quality parameters	Site	Mean	S. D.	S. E.	95% Confidence interval for mean		Minimum	Maximum
					Lower Bound	Upper Bound		
Total nitrogen (TN)	1	.763207	.2661187	.0494170	.661981	.864433	.3680	1.8000
	2	.864923	.1816130	.0251852	.814362	.915484	.5410	1.3000
	3	.700926	.2193820	.0298541	.641046	.760806	.4330	1.7500
	4	.902434	.8452613	.1161056	.669451	1.135417	.1500	3.8800
Dissolved reactive phosphorus (DRP)	1	.003172	.0015600	.0002897	.002579	.003766	.0020	.0090
	2	.007288	.0045129	.0006258	.006032	.008545	.0020	.0240
	3	.004463	.0040315	.0005486	.003363	.005563	.0020	.0290
	4	.006245	.0052802	.0007253	.004790	.007701	.0020	.0260
Total Phosphorus (TP)	1	.0262069	.01973101	.00366396	.0187016	.0337122	.00900	.11700
	2	.0314038	.01460763	.00202571	.0273371	.0354706	.01300	.09100
	3	.0197963	.02172155	.00295593	.0138675	.0257251	.00300	.16300
	4	.0272264	.05511828	.00757108	.0120339	.0424189	.00200	.35500
Flow	1	.0571893	.17753005	.02806997	.0004125	.1139662	.00000	1.11154
	2	.1945304	.21306873	.03249268	.1289575	.2601032	.00461	.99190
	3	.1601059	.61988926	.08358585	-.0074737	.3276854	.00000	4.49933
	4	.0614462	.16483072	.02913823	.0020184	.1208740	.00000	.88005
TN/TP	1	35.0239	13.54854	2.51590	29.8704	40.1775	15.38	61.20
	2	30.3887	7.66188	1.06251	28.2556	32.5217	13.19	50.62
	3	48.3366	32.61897	4.43888	39.4333	57.2398	10.74	227.33
	4	67.2065	45.74250	6.28322	54.5983	79.8147	10.65	220.00

Appendix 1.3 Robust test of equality of means for 1999-2008 data.

Robust Tests of Equality of Means					
		Statistic[a]	df1	df2	Sig.
Temperature	Welch	1.368	3	115.076	.256
Dissolved oxygen (DO)	Welch	22.193	3	111.427	.000
Turbidity	Welch	5.079	3	96.289	.003
Electrical conductivity (EC)	Welch	41.091	3	105.700	.000
pH	Welch	35.877	3	108.082	.000
Nitrate	Welch	21.128	3	92.150	.000
Total nitrogen (TN)	Welch	6.206	3	85.676	.001
Dissolved reactive phosphorus (DRP)	Welch	14.843	3	101.712	.000
Total phosphorus (TP)	Welch	3.490	3	87.329	.019
Flow	Welch	4.290	3	91.390	.007
TN and TP ratio	Welch	15.787	3	79.653	.000

a. Asymptotically F distributed.

Appendix 1.4 Multiple comparisons of means for 1999-2008 data, where 1 = Baden, 2 = Downstream of Craigbourne Dam, 3 = Richmond, 4 = White Kangaroo Rivulet.

Multiple Comparisons of Means

Games-Howell

Dependent Variable	(I) Site	(J) Site	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval Lower Bound	95% Confidence Interval Upper Bound
Temperature	1	2	-1.23977	1.04932	.640	-3.9841	1.5045
		3	-1.53665	1.08237	.490	-4.3651	1.2918
		4	-.01318	1.03267	1.000	-2.7149	2.6885
	2	1	1.23977	1.04932	.640	-1.5045	3.9841
		3	-.29688	.93984	.989	-2.7502	2.1565
		4	1.22659	.88214	.508	-1.0765	3.5297
	3	1	1.53665	1.08237	.490	-1.2918	4.3651
		2	.29688	.93984	.989	-2.1565	2.7502
		3	1.52347	.92122	.353	-.8813	3.9282
	4	1	.01318	1.03267	1.000	-2.6885	2.7149
		2	-1.22659	.88214	.508	-3.5297	1.0765
		3	-1.52347	.92122	.353	-3.9282	.8813
Dissolved oxygen (DO)	1	2	-2.39222*	.36793	.000	-3.3581	-1.4263
		3	-.76229	.41187	.256	-1.8389	.3143
		4	-.13855	.49628	.992	-1.4349	1.1578
	2	1	2.39222*	.36793	.000	1.4263	3.3581
		3	1.62994*	.31219	.000	.8136	2.4463
		4	2.25367*	.41728	.000	1.1573	3.3500
	3	1	.76229	.41187	.256	-.3143	1.8389
		2	-1.62994*	.31219	.000	-2.4463	-.8136
		4	.62373	.45649	.523	-.5703	1.8178
	4	1	.13855	.49628	.992	-1.1578	1.4349
		2	-2.25367*	.41728	.000	-3.3500	-1.1573
		3	-.62373	.45649	.523	-1.8178	.5703

*.The mean difference is significant at the 0.05 level

Appendix 1.5 Multiple comparisons of means for 1999-2008 data, where 1 = Baden, 2 = Downstream of Craigbourne Dam, 3 = Richmond, 4 = White Kangaroo Rivulet.

Multiple Comparisons of Means

Games-Howell

Dependent Variable	(I) Site	(J) Site	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
						Lower Bound	Upper Bound
Turbidity	1	2	2.52838*	.88014	.028	.2052	4.8515
		3	.19560	1.96914	1.000	-4.9770	5.3682
		4	-1.74767	1.80095	.767	-6.4740	2.9787
	2	1	-2.52838*	.88014	.028	-4.8515	-.2052
		3	-2.33278	1.81021	.574	-7.1236	2.4581
		4	-4.27604	1.62567	.052	-8.5792	.0271
	3	1	-.19560	1.96914	1.000	-5.3682	4.9770
		2	2.33278	1.81021	.574	-2.4581	7.1236
		4	-1.94327	2.39701	.849	-8.2002	4.3136
	4	1	1.74767	1.80095	.767	-2.9787	6.4740
		2	4.27604	1.62567	.052	-.0271	8.5792
		3	1.94327	2.39701	.849	-4.3136	8.2002
Electrical conductivity (EC)	1	2	-122.64092*	30.01291	.001	-201.4890	-43.7929
		3	-418.11159*	42.48260	.000	-529.0929	-307.1303
		4	-397.06580*	57.34887	.000	-547.5532	-246.5784
	2	1	122.64092*	30.01291	.001	43.7929	201.4890
		3	-295.47067*	35.91257	.000	-389.9126	-201.0288
		4	-274.42488*	52.66720	.000	-413.5426	-135.3071
	3	1	418.11159*	42.48260	.000	307.1303	529.0929
		2	295.47067*	35.91257	.000	201.0288	389.9126
		4	21.04579	60.64512	.986	-137.6578	179.7494
	4	1	397.06580*	57.34887	.000	246.5784	547.5532
		2	274.42488*	52.66720	.000	135.3071	413.5426
		3	-21.04579	60.64512	.986	-179.7494	137.6578

*.The mean difference is significant at the 0.05 level

Appendix 1.6 Multiple comparisons of means for 1999-2008 data, where 1 = Baden, 2 = Downstream of Craigbourne Dam, 3 = Richmond, 4 = White Kangaroo Rivulet.

Multiple Comparisons of Means

Games-Howell

Dependent Variable	(I) Site	(J) Site	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval Lower Bound	Upper Bound
Water pH	1	2	-.84747*	.09255	.000	-1.0896	-.6053
		3	-.37730*	.08639	.000	-.6039	-.1507
		4	-.29358*	.07937	.003	-.5031	-.0841
	2	1	.84747*	.09255	.000	.6053	1.0896
		3	.47017*	.06979	.000	.2878	.6525
		4	.55388*	.06089	.000	.3939	.7138
	3	1	.37730*	.08639	.000	.1507	.6039
		2	-.47017*	.06979	.000	-.6525	-.2878
		4	.08371	.05104	.362	-.0499	.2173
	4	1	.29358*	.07937	.003	.0841	.5031
		2	-.55388*	.06089	.000	-.7138	-.3939
		3	-.08371	.05104	.362	-.2173	.0499
Nitrate	1	2	-.0553647*	.0112939	.000	-.085274	-.025456
		3	-.0265057	.0102321	.057	-.053557	.000546
		4	-.1756630*	.0291613	.000	-.253026	-.098300
	2	1	.0553647*	.0112939	.000	.025456	.085274
		3	.0288590	.0148540	.217	-.009935	.067653
		4	-.1202983*	.0310858	.001	-.202211	-.038386
	3	4	.0265057	.0102321	.057	-.000546	.053557
		2	-.0288590	.0148540	.217	-.067653	.009935
		4	-.1491572*	.0307160	.000	-.230180	-.068134
	4	1	.1756630*	.0291613	.000	.098300	.253026
		2	.1202983*	.0310858	.001	.038386	.202211
		3	.1491572*	.0307160	.000	.068134	.230180

*.The mean difference is significant at the 0.05 level

Appendix 1.7 Multiple comparisons of means for 1999-2008 data, where 1 = Baden, 2 = Downstream of Craigbourne Dam, 3 = Richmond, 4 = White Kangaroo Rivulet.

Multiple Comparisons of Means

Games-Howell

Dependent Variable	(I) Site	(J) Site	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval Lower Bound	95% Confidence Interval Upper Bound
Total nitrogen (TN)	1	2	-.1017162	.0554647	.272	-.249962	.046530
		3	.0622810	.0577348	.704	-.091290	.215852
		4	-.1392271	.1261846	.689	-.471515	.193061
	2	1	.1017162	.0554647	.272	-.046530	.249962
		3	.1639972*	.0390584	.000	.061976	.266018
		4	-.0375109	.1188057	.989	-.351947	.276925
	3	1	-.0622810	.0577348	.704	-.215852	.091290
		2	-.1639972*	.0390584	.000	-.266018	-.061976
		3	-.2015080	.1198823	.343	-.518475	.115459
	4	1	.1392271	.1261846	.689	-.193061	.471515
		2	.0375109	.1188057	.989	-.276925	.351947
		3	.2015080	.1198823	.343	-.115459	.518475
Dissolved reactive phosphorus (DRP)	1	2	-.0041160*	.0006896	.000	-.005931	-.002301
		3	-.0012905	.0006204	.169	-.002920	.000339
		4	-.0030729*	.0007810	.001	-.005131	-.001015
	2	1	.0041160*	.0006896	.000	.002301	.005931
		3	.0028255*	.0008323	.005	.000652	.004999
		4	.0010432	.0009580	.697	-.001459	.003546
	3	1	.0012905	.0006204	.169	-.000339	.002920
		2	-.0028255*	.0008323	.005	-.004999	-.000652
		4	-.0017823	.0009094	.210	-.004160	.000595
	4	1	.0030729*	.0007810	.001	.001015	.005131
		2	-.0010432	.0009580	.697	-.003546	.001459
		3	.0017823	.0009094	.210	-.000595	.004160

*.The mean difference is significant at the 0.05 level

Appendix 1.8 Multiple comparisons of means for 1999-2008 data, where 1 = Baden, 2 = Downstream of Craigbourne Dam, 3 = Richmond, 4 = White Kangaroo Rivulet.

Multiple Comparisons of Means

Games-Howell

Dependent Variable	(I) Site	(J) Site	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval Lower Bound	95% Confidence Interval Upper Bound
Total Phosphorus (TP)	1	2	-.00519695	.00418666	.604	-.0163619	.0059680
		3	.00641060	.00470766	.528	-.0060162	.0188374
		4	-.00101952	.00841105	.999	-.0231419	.0211029
	2	1	.00519695	.00418666	.604	-.0059680	.0163619
		3	.01160755*	.00358344	.009	.0022332	.0209819
		4	.00417743	.00783739	.951	-.0165390	.0248938
	3	1	-.00641060	.00470766	.528	-.0188374	.0060162
		2	-.01160755*	.00358344	.009	-.0209819	-.0022332
		3	-.00743012	.00812765	.797	-.0288398	.0139796
	4	1	.00101952	.00841105	.999	-.0211029	.0231419
		2	-.00417743	.00783739	.951	-.0248938	.0165390
		3	.00743012	.00812765	.797	-.0139796	.0288398
Flow	1	2	-.13734103*	.04293829	.010	-.2500034	-.0246786
		3	-.10291656	.08817322	.649	-.3353425	.1295094
		4	-.00425688	.04045935	1.000	-.1107995	.1022858
	2	1	.13734103*	.04293829	.010	.0246786	.2500034
		3	.03442447	.08967925	.981	-.2016368	.2704858
		4	.13308415*	.04364413	.017	.0183297	.2478387
	3	1	.10291656	.08817322	.649	-.1295094	.3353425
		2	-.03442447	.08967925	.981	-.2704858	.2016368
		4	.09865968	.08851910	.682	-.1346315	.3319509
	4	1	.00425688	.04045935	1.000	-.1022858	.1107995
		2	-.13308415*	.04364413	.017	-.2478387	-.0183297
		3	-.09865968	.08851910	.682	-.3319509	.1346315

*.The mean difference is significant at the 0.05 level

Appendix 1.9 Multiple comparisons of means for 1999-2008 data, where 1 = Baden, 2 = Downstream of Craigbourne Dam, 3 = Richmond, 4 = White Kangaroo Rivulet.

Multiple Comparisons of Means							
Games-Howell							
Dependent Variable	**(I) Site**	**(J) Site**	**Mean Difference (I-J)**	**Std. Error**	**Sig.**	**95% Confidence Interval**	
						Lower Bound	**Upper Bound**
TN/TP	1	2	4.63528	2.73106	.339	-2.6998	11.9704
		3	-13.31261	5.10229	.052	-26.7099	.0847
		4	-32.18258*	6.76820	.000	-50.0160	-14.3492
	2	1	-4.63528	2.73106	.339	-11.9704	2.6998
		3	-17.94788*	4.56427	.001	-30.0146	-5.8812
		4	-36.81785*	6.37242	.000	-53.7009	-19.9348
	3	1	13.31261	5.10229	.052	-.0847	26.7099
		2	17.94788*	4.56427	.001	5.8812	30.0146
		4	-18.86997	7.69301	.074	-38.9921	1.2521
	4	1	32.18258*	6.76820	.000	14.3492	50.0160
		2	36.81785*	6.37242	.000	19.9348	53.7009
		3	18.86997	7.69301	.074	-1.2521	38.9921

*.The mean difference is significant at the 0.05 level

Appendix 2.1 Summary statistics for 2005/7data, where 1= Baden, 2 = Downstream of Craigbourne Dam, 3 = Richmond, 4 = White Kangaroo Rivulet.

Water quality parameters	Site	Mean	S.D	S.E	95% Confidence Interval for Mean		Minimum	Maximum
					Lower Bound	Upper Bound		
Temperature	1	9.8313	5.30481	1.32620	7.0045	12.6580	3.50	21.00
	2	12.4500	4.13383	.97435	10.3943	14.5057	6.70	19.80
	3	13.4000	5.47449	1.25593	10.7614	16.0386	6.20	27.10
	4	11.5842	4.59544	1.05427	9.3693	13.7991	3.30	21.30
Dissolved oxygen (DO)	1	7.5638	1.75611	.43903	6.6280	8.4995	4.69	10.16
	2	11.0856	1.25620	.29609	10.4609	11.7103	8.73	12.82
	3	9.6247	1.38993	.31887	8.9548	10.2947	6.60	11.54
	4	9.2242	1.73164	.39727	8.3896	10.0588	5.70	12.20
Water pH	1	7.2706	.63088	.15772	6.9345	7.6068	6.39	8.87
	2	8.1344	.39927	.09411	7.9359	8.3330	7.06	8.72
	3	7.8095	.43845	.10059	7.5981	8.0208	6.80	8.44
	4	7.7047	.18563	.04259	7.6153	7.7942	7.33	7.97
Electrical conductivity (EC)	1	530.0000	226.72715	56.68179	409.1856	650.8144	219.00	1051.00
	2	525.2778	76.19542	17.95943	487.3867	563.1689	380.00	692.00
	3	893.8421	243.63207	55.89303	776.4152	1011.2690	503.00	1327.00
	4	1055.4211	367.32884	84.27102	878.3742	1232.4679	315.00	1718.00
Turbidity	1	4.7056	4.66885	1.16721	2.2178	7.1935	1.06	19.30
	2	2.3711	1.51449	.35697	1.6180	3.1243	1.08	6.70
	3	2.8263	4.42027	1.01408	.6958	4.9568	.63	20.10
	4	4.2537	2.14596	.49232	3.2194	5.2880	1.91	9.30

Appendix 2.2 Summary statistics for 2005/7data, where 1= Baden, 2 = Downstream of Craigbourne Dam, 3 = Richmond, 4 = White Kangaroo Rivulet.

Water quality parameters	**Site**	**Mean**	**S.D.**	**S.E.**	**95% Confidence Interval for Mean**		**Minimum**	**Maximum**
					Lower Bound	Upper Bound		
Dissolved reactive phosphorus (DRP)	1	.002875	.0017464	.0004366	.001944	.003806	.0020	.0090
	2	.005056	.0020714	.0004882	.004025	.006086	.0020	.0100
	3	.002895	.0009941	.0002281	.002416	.003374	.0020	.0060
	4	.003684	.0011572	.0002655	.003126	.004242	.0020	.0060
Total phosphorus (TP)	1	.0204375	.00970889	.00242722	.0152640	.0256110	.00900	.03600
	2	.0224444	.00624238	.00147134	.0193402	.0255487	.01600	.03800
	3	.0154737	.00537865	.00123395	.0128813	.0180661	.00500	.02800
	4	.0080526	.00311758	.00071522	.0065500	.0095553	.00500	.01500
Nitrate	1	.004625	.0097151	.0024288	-.000552	.009802	.0020	.0410
	2	.020833	.0269995	.0063638	.007407	.034260	.0020	.1160
	3	.017211	.0327678	.0075175	.001417	.033004	.0020	.1490
	4	.274684	.2478458	.0568597	.155226	.394142	.0020	.7570
Total nitrogen (TN)	1	.694875	.1746787	.0436697	.601795	.787955	.3820	1.0000
	2	.712222	.1055805	.0248856	.659718	.764726	.5410	.9300
	3	.625053	.0746045	.0171154	.589094	.661011	.5200	.8280
	4	.638053	.2746461	.0630081	.505677	.770428	.2830	1.2000

Appendix 2.3 Robust test of equality of means for 2005/7data.

Robust Tests of Equality of Means					
		Statistic[a]	df1	df2	Sig.
Temperature	Welch	1.373	3	37.097	.266
Dissolved oxygen(DO)	Welch	15.132	3	36.879	.000
Water pH	Welch	8.770	3	32.938	.000
Electrical conductivity(EC)	Welch	23.463	3	31.835	.000
Turbidity	Welch	3.731	3	34.380	.020
Dissolved reactive phosphorus(DRP)	Welch	6.086	3	35.470	.002
Total phosphorus(TP)	Welch	32.283	3	33.960	.000
Nitrate	Welch	9.452	3	33.178	.000
Total nitrogen(TN)	Welch	2.967	3	34.228	.056

a. Asymptotically F distributed.

Appendix 2.4 Multiple comparisons of means for 2005/7 data, where 1= Baden, 2 = Downstream of Craigbourne Dam, 3 = Richmond, 4 = White Kangaroo Rivulet.

Multiple Comparisons of Means

Games-Howell

Dependent Variable	(I) Site	(J) Site	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
						Lower Bound	Upper Bound
Temperature	1	2	-2.61875	1.64565	.399	-7.1090	1.8715
		3	-3.56875	1.82652	.226	-8.5149	1.3774
		4	-1.75296	1.69419	.731	-6.3599	2.8540
	2	1	2.61875	1.64565	.399	-1.8715	7.1090
		3	-.95000	1.58957	.932	-5.2471	3.3471
		4	.86579	1.43556	.930	-3.0063	4.7378
	3	1	3.56875	1.82652	.226	-1.3774	8.5149
		2	.95000	1.58957	.932	-3.3471	5.2471
		4	1.81579	1.63977	.687	-2.6068	6.2384
	4	1	1.75296	1.69419	.731	-2.8540	6.3599
		2	-.86579	1.43556	.930	-4.7378	3.0063
		3	-1.81579	1.63977	.687	-6.2384	2.6068
Dissolved oxygen (DO)	1	2	-3.52181*	.52954	.000	-4.9715	-2.0722
		3	-2.06099*	.54261	.004	-3.5412	-.5808
		4	-1.66046*	.59209	.040	-3.2651	-.0558
	2	1	3.52181*	.52954	.000	2.0722	4.9715
		3	1.46082*	.43514	.010	.2872	2.6345
		4	1.86135*	.49547	.004	.5208	3.2019
	3	1	2.06099*	.54261	.004	.5808	3.5412
		2	-1.46082*	.43514	.010	-2.6345	-.2872
		4	.40053	.50941	.860	-.9745	1.7756
	4	1	1.66046*	.59209	.040	.0558	3.2651
		2	-1.86135*	.49547	.004	-3.2019	-.5208
		3	-.40053	.50941	.860	-1.7756	.9745

*. The mean difference is significant at the 0.05 level.

Appendix 2.5 Multiple comparisons of means for 2005/7 data, where 1= Baden, 2 = Downstream of Craigbourne Dam, 3 = Richmond, 4 = White Kangaroo Rivulet.

Multiple Comparisons of Means

Games-Howell

Dependent Variable	(I) Site	(J) Site	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval Lower Bound	95% Confidence Interval Upper Bound
Water pH	1	2	-.86382*	.18366	.000	-1.3693	-.3584
		3	-.53885*	.18706	.037	-1.0519	-.0258
		4	-.43411	.16337	.071	-.8980	.0297
	2	1	.86382*	.18366	.000	.3584	1.3693
		3	.32497	.13775	.104	-.0465	.6965
		4	.42971*	.10330	.002	.1445	.7149
	3	1	.53885*	.18706	.037	.0258	1.0519
		2	-.32497	.13775	.104	-.6965	.0465
		4	.10474	.10923	.774	-.1964	.4058
	4	1	.43411	.16337	.071	-.0297	.8980
		2	-.42971*	.10330	.002	-.7149	-.1445
		3	-.10474	.10923	.774	-.4058	.1964
Electrical conductivity(EC)	1	2	4.72222	59.45895	1.000	-163.3234	172.7679
		3	-363.84211*	79.60437	.000	-579.2925	-148.3917
		4	-525.42105*	101.55998	.000	-801.3219	-249.5202
	2	1	-4.72222	59.45895	1.000	-172.7679	163.3234
		3	-368.56433*	58.70751	.000	-531.7856	-205.3430
		4	-530.14327*	86.16348	.000	-771.7076	-288.5789
	3	1	363.84211*	79.60437	.000	148.3917	579.2925
		2	368.56433*	58.70751	.000	205.3430	531.7856
		4	-161.57895	101.12188	.394	-435.8998	112.7420
	4	1	525.42105*	101.55998	.000	249.5202	801.3219
		2	530.14327*	86.16348	.000	288.5789	771.7076
		3	161.57895	101.12188	.394	-112.7420	435.8998

*. The mean difference is significant at the 0.05 level.

Appendix 2.6 Multiple comparisons of means for 2005/7 data, where 1= Baden, 2 = Downstream of Craigbourne Dam, 3 = Richmond, 4 = White Kangaroo Rivulet.

Multiple Comparisons of Means

Games-Howell

Dependent Variable	(I) Site	(J) Site	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval Lower Bound	95% Confidence Interval Upper Bound
Turbidity	1	2	2.33451	1.22058	.258	-1.1190	5.7880
		3	1.87931	1.54620	.622	-2.3148	6.0734
		4	.45194	1.26679	.984	-3.0895	3.9934
	2	1	-2.33451	1.22058	.258	-5.7880	1.1190
		3	-.45520	1.07507	.974	-3.4366	2.5262
		4	-1.88257*	.60811	.020	-3.5290	-.2361
	3	1	-1.87931	1.54620	.622	-6.0734	2.3148
		2	.45520	1.07507	.974	-2.5262	3.4366
		4	-1.42737	1.12727	.592	-4.5195	1.6648
	4	1	-.45194	1.26679	.984	-3.9934	3.0895
		2	1.88257*	.60811	.020	.2361	3.5290
		3	1.42737	1.12727	.592	-1.6648	4.5195

*. The mean difference is significant at the 0.05 level.

Appendix 2.7 Multiple comparisons of means for 2005/7 data, where 1= Baden, 2 = Downstream of Craigbourne Dam, 3 = Richmond, 4 = White Kangaroo Rivulet.

Multiple Comparisons of Means

Games-Howell

Dependent Variable	**(I) Site**	**(J) Site**	**Mean Difference (I-J)**	**Std. Error**	**Sig.**	**95% Confidence Interval**	
						Lower Bound	**Upper Bound**
Dissolved reactive phosphorus (DPR)	1	2	-.0021806*	.0006550	.011	-.003955	-.000406
		3	-.0000197	.0004926	1.000	-.001383	.001344
		4	-.0008092	.0005110	.405	-.002214	.000595
	2	1	.0021806*	.0006550	.011	.000406	.003955
		3	.0021608*	.0005389	.003	.000675	.003647
		4	.0013713	.0005558	.089	-.000152	.002895
	3	1	.0000197	.0004926	1.000	-.001344	.001383
		2	-.0021608*	.0005389	.003	-.003647	-.000675
		4	-.0007895	.0003500	.128	-.001733	.000154
	4	1	.0008092	.0005110	.405	-.000595	.002214
		2	-.0013713	.0005558	.089	-.002895	.000152
		3	.0007895	.0003500	.128	-.000154	.001733
Total Phosphorus (TP)	1	2	-.00200694	.00283835	.893	-.0098129	.0057990
		3	.00496382	.00272287	.289	-.0025838	.0125114
		4	.01238487*	.00253040	.001	.0052176	.0195521
	2	1	.00200694	.00283835	.893	-.0057990	.0098129
		3	.00697076*	.00192028	.005	.0017815	.0121600
		4	.01439181*	.00163597	.000	.0098878	.0188958
	3	1	-.00496382	.00272287	.289	-.0125114	.0025838
		2	-.00697076*	.00192028	.005	-.0121600	-.0017815
		4	.00742105*	.00142624	.000	.0035342	.0113079
	4	1	-.01238487*	.00253040	.001	-.0195521	-.0052176
		2	-.01439181*	.00163597	.000	-.0188958	-.0098878
		3	-.00742105*	.00142624	.000	-.0113079	-.0035342

*. The mean difference is significant at the 0.05 level.

Appendix 2.8 Multiple comparisons of means for 2005/7 data, where 1= Baden, 2 = Downstream of Craigbourne Dam, 3 = Richmond, 4 = White Kangaroo Rivulet.

Multiple Comparisons of Means							
Games-Howell							
						95% Confidence Interval	
Dependent Variable	**(I) Site**	**(J) Site**	**Mean Difference (I-J)**	**Std. Error**	**Sig.**	**Lower Bound**	**Upper Bound**
Nitrate	1	2	-.0162083	.0068116	.111	-.035137	.002721
		3	-.0125855	.0079001	.403	-.034549	.009378
		4	-.2700592*	.0569116	.001	-.430851	-.109268
	2	1	.0162083	.0068116	.111	-.002721	.035137
		3	.0036228	.0098494	.983	-.022964	.030210
		4	-.2538509*	.0572148	.002	-.415172	-.092530
	3	1	.0125855	.0079001	.403	-.009378	.034549
		2	-.0036228	.0098494	.983	-.030210	.022964
		4	-.2574737*	.0573545	.001	-.419042	-.095905
	4	1	.2700592*	.0569116	.001	.109268	.430851
		2	.2538509*	.0572148	.002	.092530	.415172
		3	.2574737*	.0573545	.001	.095905	.419042

*. The mean difference is significant at the 0.05 level.